Rice Lake Water Quality Before and After Reactivation of the San Antonio Gold Mine

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RICE LAKE WATER QUALITY BEFORE AND AFTER REACTIVATION OF THE SAN ANTONIO GOLD MINE

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ABSTRACT

Studies conducted prior to reactivation of the San Antonio mine and mill determined that historic mine operation had loaded iron, manganese, copper and mercury to the sediments of Rice Lake. Site preparatory discharges of mine shaft water to Rice Lake caused significant elevations in water column conductivity, pH, alkalinity, calcium, lead and copper. Elevated fish muscle tissue levels of copper and lead were attributed to shaft water discharges while elevated mercury levels were attributed to historic sediment loading. Long term impacts upon benthos and phytoplankton were not detected. Neither a minor tailings spill shortly after start up nor five months operation of the reactivated mine and mill caused any detectable impacts upon water physicochemistry.

SUMMARY

A proposal to reactivate the San Antonio mine and mill raised concerns about environmental impact of the operation upon Rice Lake. A pre-reactivation study was conducted in order to develop an extensive background data base from which potential future impacts could be assessed. Water physicochemistry, metal levels in sediments and fish muscle tissue, benthos populations and phytoplankton were examined. A spill of tailings shortly after start up prompted a second study. Efforts in this instance were restricted to assessing impact upon the water column. A third study was conducted approximately five months after start up in order to assess initial impacts upon water physicochemistry.

The pre-reactivation study determined that historic operation of the mine and mill had resulted in significant loading of mercury, copper, iron and manganese to the sediments of Rice Lake. Iron and manganese appeared to have been physically eroded from the tailings area whereas mercury and copper appeared to have been transported by cyanide complexation. Elevated mercury concentrations in fish muscle tissue were attributed to historic sediment loading. Long term negative impacts upon benthos and phytoplankton were not detectable.

The pre-reactivation study also determined that mine shaft dewatering added 0.93% to the volume of Rice Lake. This process caused a temporary significant elevation in water column levels of conductivity, pH, alkalinity, calcium, lead and copper. Elevated copper and lead concentrations in fish muscle tissue were also attributed to the dewatering process.

The January 27, 1982 spill of tailings to the ice of Rice Lake caused no detectable impact upon the water physicochemistry of Rice Lake. This was because tailings materials were frozen to the lake ice and subsequently dispersed slowly during breakup. Similarly, with the possible exception of water column arsenic concentrations, five months operation of the mine and mill caused no detectable impacts upon water physicochemistry. However, subtle impacts may have been masked by the pre-reactivation discharges of mine shaft water.

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INTRODUCTION

Brinex Limited and New Forty-Four Mines Limited undertook a joint venture in August, 1980 to evaluate the San Antonio Gold mine property at Bissett (51°02', 95°41'). A favourable feasibility study led to developmental work and subsequent reactivation of the mine and mill by January, 1982. The mine, mill and tailings disposal sites are located along the north shore of Rice Lake (51°01', 95°40'). Rice Lake is the source of potable water supply for the community of Bissett.

Concerns regarding potential environmental impacts are normally associated with prospective mining ventures. When prospective ventures involve sites near aquatic systems, many of the concerns focus upon impacts to existing aquatic conditions and the potential ramifications for users of the aquatic systems.

Water Standards and Studies Section conducted a two phase study in order to address aquatic concerns. The first phase involved development of an extensive background (pre-reactivation) data base from which potential future impacts could be measured. The second phase involved limited post-reactivation data from which an assessment of initial impacts could be made. Specific objectives established to achieve these goals included the following:

a) Determination of pre-reactivation status of Rice Lake respecting:
   - water physicochemistry;
   - sediment metal concentrations,
   - fish muscle tissue metal concentrations,
   - plankton and benthic population characteristics;

b) Determination of post-reactivation Rice Lake water quality status during:

- ice cover conditions,
- open water conditions;

c) Assessment of initial reactivation impacts upon Rice Lake water quality.

AREA DESCRIPTION

The study area lies approximately 160 kilometres north-east of Winnipeg (Figure 1).

Rice Lake Environs

Physiographically the study area falls within the Precambrian Drift Plain subregion (Weir, 1961). Varying thicknesses of glacial drift overlie much of the precambrian bedrock in the area (Ibid). However, outcrops are frequent and are often interspersed with bogs. This gives rise to a moderately hilly topography. Soils are typically poor and consist of greywooded and podzols. Black spruce and tamarack communities dominate lower sites whereas deciduous trees and jackpine colonize well drained sites and outcrops.

Feldspathic quartizite and conglomerate formations known collectively as the San Antonio formation border the western side of Rice Lake (Davies et al, 1963). The Rice Lake Group of volcanics surround the remainder of the Lake (Ibid). Brinco Mining Limited (1981) presents a detailed description of regional and mine specific geology.

Terrain immediately north of Rice Lake is dominated by an outcrop upon which the mine site and much of the community of Bissett are established (Figure 1). Westerly, the topography is flat and consists of a low lying open bog. Southerly, the terrain is hilly and heavily forested. Terrain immediately north-east of Rice Lake is flat and lowlying. Unlike the western area, the eastern bog has a nearly closed canopy.

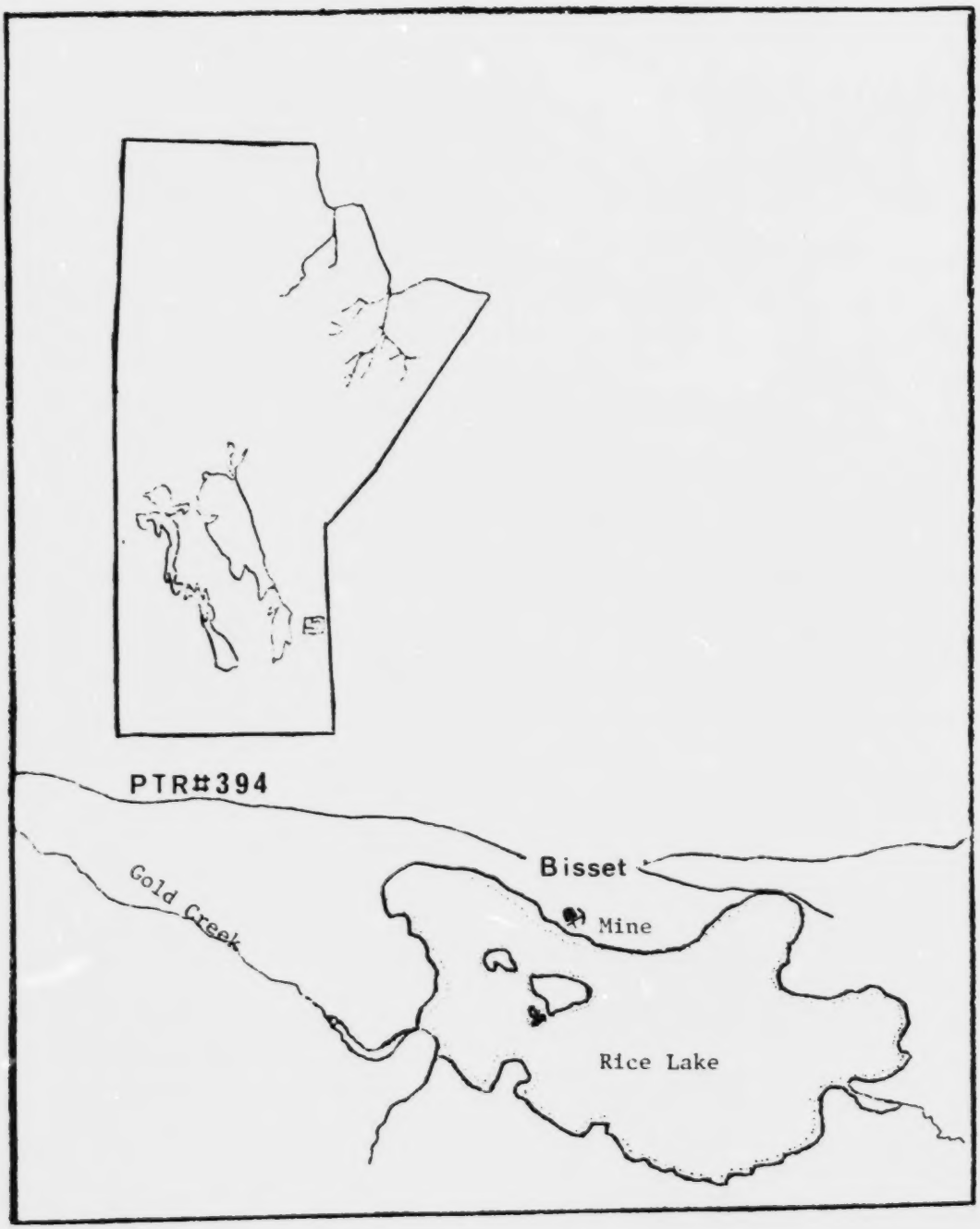

Figure 1: Overview of the Study Area.

Rice Lake Morphometry and Bathymetry

Rice Lake is an oblong shaped lake oriented in a northwest-southeast direction. Maximum direct length and breadth are approximately 3.45 kilometres and 1.9 kilometres, respectively. The lake has a surface area of 3.82 square kilometres and has a mean depth of 2.55 metres. The maximum depth encountered during the present study was 5.1 metres (Figure 2). Volume at time of survey (June 2, 1982) was estimated at 9.75×10^6 cubic metres. The bathymetry is illustrated in Figure 2.

Seven minor islands are scattered about the lake. These are little more than exposed reefs and are lightly vegetated. Two larger islands, approximately 3.5 hectares in total area, are located near the mine site.

Rice Lake is a first order lake of the Wanipigow River system and receives drainage from approximately 52 square kilometres of watershed. Permanent influent streams enter the lake at its easterly and southerly margins (Gold and Red Rice Creeks, respectively). Two intermittent streams, Independence Creek (easterly margin) and an unamed creek (westerly margin) also discharge periodically to Rice Lake. Gold Creek drains the lake from its mid-southerly margin (Figure 2). Water levels are maintained by a log-rock-earthen dam located approximately 0.75 kilometres downstream of Rice Lake.

Bottom sediments over much of the lake consist of greyish silts and organic debris tightly bound by the roots of aquatic macrophytes. Plant debris content generally increases with proximity to shorelines. Tailings comprise the sole sediment type in the

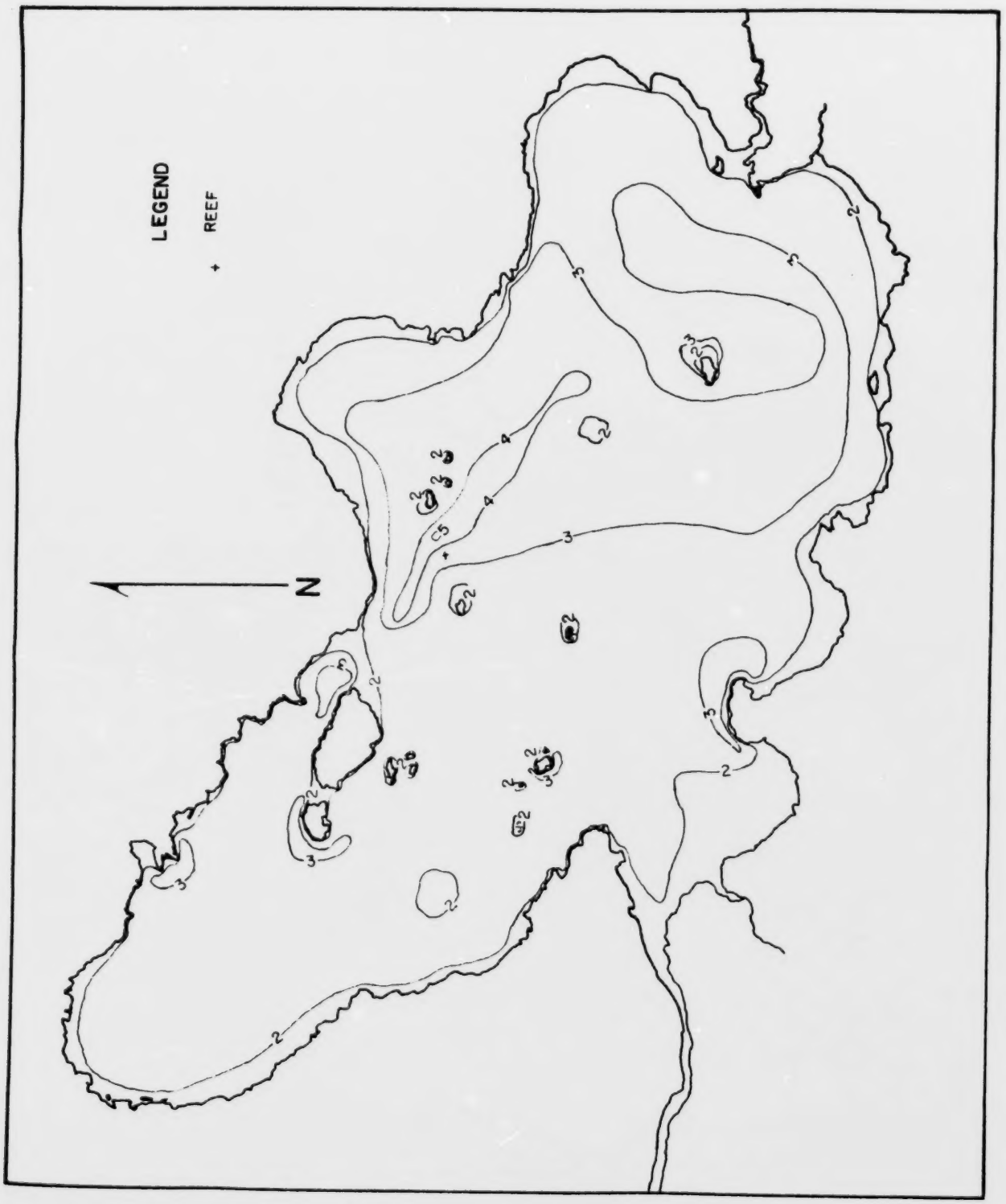

Figure 2: Contour map of Rice Lake.

immediate vicinity of the original tailings disposal area. These inundated tailings have been lightly colonized by Scirpus sp. Emergent beds of Scirpus sp. also occur at middle-southerly locations on the lake. Extensive submergent beds of Myriophyllum sp. and various Potomogeton spp. are abundant throughout the lake.

SUMMARY OF MINE OPERATIONS

San Antonio Gold Mine*

The first important discovery of gold in Manitoba was made at Rice Lake. The San Antonio and adjoining claims were staked early in 1911 by E. A. Pelletier and A. Desoutels. However, little development work was done on the properties until after World War I.

The Wanipigow Syndicate was formed in 1926 to explore the San Antonio and adjoining claims. Shaft sinking began in 1927. Later that year the name was changed to San Antonio Mines Limited. A further name change occurred in 1931 when the property was refinanced and incoporated as San Antonio Gold Mines Limited. Mine production commenced in 1932 and continued uninterrupted until 1966. The mill continued operating until 1968. The operation was then closed and placed in receivership due to a major fire and increasing financial difficulties.

The deposit was located in the "greenstone" belt on the north shore of Rice Lake. The gold was borne in a network of quartz veins and stringers bounded by wall rocks that were intensely pyritized. Including output from leased Forty-Four Mines, a total of 3,983,035 tonnes of ore were milled. A total of 36,652.5 kilograms of gold and 5,859.9 kilograms of silver were recovered.

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*Summarized from Mineral Development Sector, 1974.

The Joint Venture*

New Forty-Four Mines Limited owned a large block of mineral claims in the Rice Lake area which comprised part of the former San Antonio Gold Mine. Brinex Limited signed a Joint Venture Agreement with New Forty-Four Mines in August, 1980. This agreement provided for joint evaluation and development of the property.

Following the agreement, Brinex assumed responsibility for mine dewatering and ore reserve confirmation. An encouraging preliminary "Feasibility Study" was published in September, 1980. Much of the engineering design work was completed by the end of January, 1981. Considerable minesite reconditioning work was also completed. This included dewatering to the 31st level and the subsequent discharge of 90,912 cubic metres of minewater directly to Rice Lake.

Upon completion of the final feasibility study in mid February, 1981, the decision was made to reactivate the mine, construct a new concentrator and rehabilitate mill facilities. Mill capacity was designed to treat 110,677 tonnes of ore per year and produce an estimated 715.3 kilograms of gold per year. Mineable ore reserves were estimated at 740,668 tonnes grading 5.91 grams gold per tonne of ore. Both the mine and the mill were in production by January, 1982 (R. Johns, pers. comm.).

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*Summarized from Brinco Mining Limited, 1981.

Mill tailings were to be discharged to an impoundment area located atop the former San Antonio tailings site. Water from the tailings pond was to be reclaimed for use in the mill process. However, discharge of pond supernate to Rice Lake occurred during an impoundment dam failure on January 27, 1982. The incident resulted in an estimated discharge of 45 to 68 cubic metres of supernate to the frozen surface of Rice Lake (D. Peterson, pers. comm.). Seepage from the impoundment itself was expected to be minimal due to the relatively impervious nature of the original tailings.

STUDY METHODS AND MATERIALS

Study Design

The study was initially designed to delineate Rice Lake water quality conditions existing prior to reactivation of the mine and mill and to assess initial impacts shortly after reactivation.
Two sampling periods were planned. The intensive pre reactivation baseline study was conducted May 19-21, 1981. The impact assessment which concentrated on water physicochemistry was conducted during open water conditions on May 31 - June 2, 1982. However, an inadvertent tailings spill on January 27, 1982 prompted an intermediate ice cover sampling session on March 23-24, 1982. Bathymetric characteristics of Rice Lake were determined during the 1982 open water session.

Due to the relatively small size and shallow depth of Rice Lake, a total of fifteen sampling stations were considered adequate for the definition of water column physicochemistry and sediment metal characteristics within statistically significant limits. The sampling stations were randomly identified (Figure 3) by superimposing a numbered grid upon a diagram of the lake and utilizing random number tables to detemine grid reference points.

Field Procedures

Water samples were collected in a similar manner during each of the three sampling periods. A four litre acrylic (trace metal grade) Kemmerer sampler was used to collect water at random depths

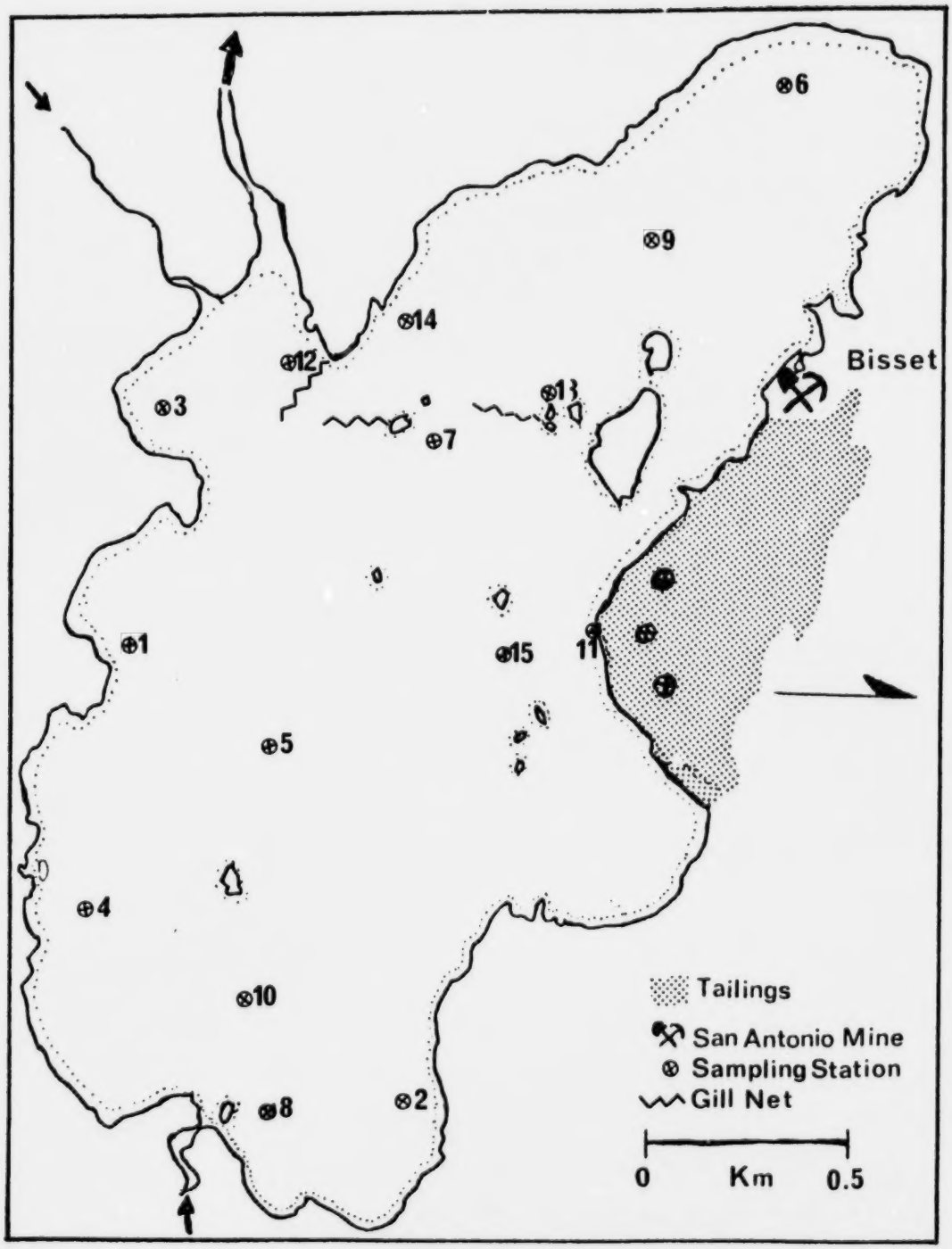

Figure 3: Rice Lake Sampling Stations and gill net sites.

from each of the randomly identified stations.* Random depths were determined by considering random numbers as a percentage of the total depth of each station. (During the ice cover sampling session, total depth was considered as extending from the lake floor to the bottom of the ice). Volumes collected were sufficient for laboratory analysis of the requested parameters. Essential preservatives were added where required immediately following collection of each sample. Water temperatures and conductivities were measured on site by utilizing a Yellow Springs Instruments model 33 conductivity meter. A Canlab model H-5503-1 pH meter was used to determine field pH values. Because of the absence of scheduled bus service to Bissett, collection of perishable water samples was relegated to the mornings of the days of departure (per sampling period) from the study area. Thus, for each sampling period, delivery of perishables to the W. M. Ward Technical Services Laboaratory (Winnipeg) by the study crew was assurred within twenty-four hours of collection.

Bathymetric data was accumulated during the open water post-reactivation session. Continuous chart recordings of depth were obtained by using a Furuno FG200 Mark 3 echo sounder with hull mounted transducer on a series of thirty-three point to point transects.

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* Due to suspected mercury contamination of the Kemmerer sampler during the pre-reactivation survey, mercury sample bottles were filled directly by submersion during both post-reactivation sampling periods.

Sediments from each station were collected during the baseline survey by using a 22.9 centimetre square Ekman dredge. Upon hoisting the Ekman grab to the lake surface, the top of the dredge was carefully opened so as to avoid disturbance of the delicate water-sediment interface. A five centimetre diameter clear acrylic tube was then inserted through the dredge contents. Excess water and sediments more than five centimetres below the interface were carefully drained from the tube. The remaining contents of the tube were then drained into a labelled "whirlpac". All sediment samples were placed in a local freezer within six hours of collection and were maintained in a frozen state until submission to the W. M. Ward Technical Services Laboratory.

The tailings disposal area was also inspected during the baseline survey. Three surface samples of tailings were collected from various locations on the site (Figure 3). These were placed in a local freezer and maintained in a frozen state until submission to the W. M. Ward Technical Services Laboratory.

Fish were collected during the baseline survey by overnight deployment of 76, 95 and 105 mm mesh (x 1.8 m deep) gill net gangs set at suitable locations within the lake (Figure 3). The total catch was identified to species and tallied. A maximum of forty specimens per species, each representative of the size range of pike, common sucker and perch collected, were then selected for sampling. Each individual was assigned an identifying number, weighed to the nearest 10 grams, measured to the nearest 0.1 centimetres and sexed. Evidences of age, consisting of either otoliths, cleithra or pelvic fin rays (species

dependent), were removed. Specimens were then filleted and the lateral muscle tissue was inserted in labelled "whirlpacs". All tissue samples were placed in a local freezer upon collection and were maintained in a frozen state until submission to the W. M. Ward Technical Services Laboratory.

Benthic samples were also collected during the baseline survey. A 22.9 centimetre square Ekman grab was taken at each station (Figures 3). The contents of each grab were placed in nitex nylon bags of 0.4 mm mesh and repeatedly dipped into the lake so as to remove finer silts and concentrate the benthic organisms. Dredge contents thus concentrated were placed in labelled "whirlpacs" and were preserved by appropriate volumes of the vital stain, rose bengal (Mason and Yevich, 1967).

Phytoplankton samples taken during the baseline survey, were collected with the aid of a 2.0 litre clear acrylic Kemmerer sampler. Samples were collected from random depths and sieved through a 64 μ mesh Wisconsin plankton bucket. Resultant concentrates were drained to labelled vials and preserved by the addition of several drops of Lugol's iodine solution. An additional phytoplankton sample was collected and retained "live" to facilitate identification of protista.

Analytical Procedures

Immediately upon receipt by the W. M. Ward Technical Services Laboratory, water samples were refrigerated and frozen samples of tissue, sediments and tailings were stored in freezers. All samples were analyzed prior to shelf-life expiration according to methods described by Sorba et al (1980).

Benthic and plankton samples were also submitted to the W. M. Ward Technical Services Laboratory for analysis. Benthic samples were washed again in the laboratory to remove fine silt materials. Organisms were then manually picked from surrounding debris with the aid of a variable 1-4X binocular dissecting microscope. Due to the great analytical expenses incurred by sorting through the large quantities of plant debris and other detritus only samples from stations 1 and 11 were analysed. Benthic organisms were identified to the lowest possible taxonomic level and ennumerated. Phytoplankters were identified to species (where possible) and both surface areas (μ^2) and numbers per litre were calculated.

Fish evidences of age were submitted for determination to the Biological Services Branch of the Manitoba Department of Natural Resources.

Continuous chart recordings of depth were interpreted by Water Standards and Studies Section staff. Depths were plotted on transect lines and contours were developed manually.

Upon completion of analyses, all data were tabulated and summarized. Analysis of variance was performed on water physicochemical data. Regressions of the various sediment metal concentrations on proximity to the tailings storage site were calculated and tested for significance of correlations. Student's "t" test was used to compare Rice Lake fish muscle tissue concentrations to those of similar species from other lakes.

RESULTS AND DISCUSSION

Water Physicochemistry

Water physicochemistry data for each sampling session are detailed in Appendices 1-3, respectively. Summaries of each session are presented in Tables 1 to 3, respectively. Results of analysis of variance are detailed in Table 4.

a) Pre-Reactivation

The pre-reactivation thermal regime of Rice Lake was unstratified as all water temperatures recorded were 18.0°C. Surface to bottom thermal differentials during all seasons were expected to be minimal due to the shallow mean depth.

The mean conductivity (340 uS/cm) was considerably greater than anticipated for lakes in a similar geophysical setting and suggested anthropogenic impact. One inference was that elevated conductivities were due to ions emanating from the tailings disposal site. However, a regression associating conductivity with proximity to the tailings site was not significantly correlated ($r^2 = 0.02$). This suggested that active transport of ions from the tailings site to the lake was not occurring at a significant rate. Elevated conductivities could also have been a product of previous mining activity which involved direct drainage of operational wastewaters from the tailings disposal area to the lake. However, previous mining activity was not implicated as raw water samples collected on November 5, 12 and 26, 1980 from the Bissett water treatment plant

showed a mean conductivity of 151 ± 13 uS/cm (Moche, 1981). Near completion of the mine dewatering process, December 15, 1980 conductivity in raw water at the Bissett water treatment plant measured 690 uS/cm (Ibid). The close association between observed mean conductivity (340 uS/cm) and estimated conductivity following complete lake dilution of mine dewatering discharges (318 uS/cm)* suggests that the elevated lake conductivity observed was due to mine dewatering discharges.

Rice Lake was non turbid as the bottom was readily visible at many sampling stations. Mean turbidity and non filterable residue values were low (1.03 NTu, and <5.0 mg/L; respectively). Anthropogenic impact was not evident.

The concentration of nutrients was considered within the range common to shallow lakes in a precambrian setting. Mean total phosphorus and mean dissolved nitrate-nitrite nitrogen concentrations (0.03 mg/L, and 0.04 mg/L, respectively) suggested a trophic status bordering on eutrophy (Vollenweider, 1968). Due to the shallow mean depth and low turbidity, rooted aquatic macrophytes were expected to proliferate and the development of nuisance algal blooms was not anticipated. Anthropogenic impact was not evident.

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* Estimated conductivity following complete lake dilution of mine dewatering discharges:

= Background conductivity + (discharge conductivity (1) X Dilution factor(2))
= 151 uS/cm + (17950 uS/cm X 0.0093)
= 318 uS/cm

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(1) Discharge conductivity = 17950 ± 12872 uS/cm (Moche, 1981).

(2) Dilution factor = Volume of discharge/Volume of Rice Lake
= 9.09×10^4 m^3/9.75×10^6 m^3
= 0.0093

TABLE 1: SUMMARY OF PRE-REACTIVATION RICE LAKE WATER PHYSICOCHEMISTRY.

PARAMETERS	STATISTIC:	MEAN	STANDARD DEVIATION	GREATEST VALUE	SMALLEST VALUE	NUMBER OF SAMPLES
Field Temperature	(°C)	18.0	0.0	18.0	18.0	15
Conductivity	(uS/cm)	340.	6.65	351.	329.	15
Residue Non Filt.	(mg/L)	<5.0	0.0	<5.0	<5.0	15
pH (log.$\bar{x}$)	(pH Units)	7.86	--	7.90	7.67	15
Alkalinity Total ($CaCO_3$)	(mg/L)*	51.0	0.91	52.	48.	14
Alkalinity (HCO_3)	(mg/L)	62.	1.01	63.	59.	15
Alkalinity (CO_3)	(mg/L)*	0.0	0.0	0.0	0.0	14
Alkalinity (OH)	(mg/L)	0.0	0.0	0.0	0.0	15
Dissolved Oxygen	(mg/L)	9.0	0.38	9.7	8.5	15
T.O.C.	(mg/L)	17.9	0.43	18.5	17.5	15
T.I.C.	(mg/L)	12.0	0.8	14.0	10.5	15
Turbidity	(NTu)	1.0	0.13	1.5	1.0	15
Kjeldhal Total-N	(mg/L)	0.6	0.05	0.6	0.5	15
NO_3-NO_2-N (Diss.)	(mg/L)	0.04	0.034	0.14	<0.01	15
P-Total	(mg/L)	0.03	0.015	0.08	0.02	15
Cyanide (Total)	(mg/L)	<0.02	0.0	<0.02	<0.02	15
As (Total)	(mg/L)	<0.001	0.0	<0.001	<0.001	15
Cd (Ext.)	(mg/L)	<0.005	0.0	<0.005	<0.005	15
Cu (Ext.)	(mg/L)	0.04	0.02	0.095	0.02	15
Pb (Ext.)	(mg/L)	0.005	0.001	0.010	<0.005	15
Ni (Ext.)	(mg/L)	0.010	0.005	0.020	<0.005	15
Zn (Ext. Dir.)	(mg/L)	0.021	0.003	0.03	0.02	15
Ca (Ext.)	(mg/L)	30.	0.72	30.	28.	15
Mg (Ext.)	(mg/L)	5.0	0.0	5.0	5.0	15
Hg (Total)	(µg/L)	<0.02	0.0	<0.02	<0.02	15

*STATION 13 VALUES HAVE BEEN EXCLUDED FROM THE CALCULATIONS ON THE BASIS OF DICKSON'S OUTLIER TEST.

Mean dissolved oxygen concentrations (9.0 mg/L) indicated a 95% level of saturation. This is consistent with the proliferation of rooted macrophytes observed.

Rice Lake is situated at the southern extremity of an area considered highly sensitive to acid precipitation. Lakes of the area are characterized by low pH values and little buffering capacity. The Rice Lake mean logarithmic pH (7.86 units) was somewhat greater than expected. However, the mean total and bicarbonate alkalinities (51. mg/L, and 62 mg/L; respectively) were moderately low and not atypical of the area. Mean calcium and magnesium concentrations (30.0 mg/L and 5.0 mg/L, respectively) were also within the expected range. However, like the pH levels, some of the physicochemical parameters capable of buffering acidity appeared to show anthropogenic impacts when compared to Rice Lake raw water samples collected at the Bissett water treatment plant prior to completion of mine shaft dewatering. Calcium concentrations appeared most affected as shaft water concentrations averaging 1894 mg/L (Moche, 1981) were much greater than raw water concentrations of 20 mg/L (Ibid). Estimated calcium concentrations* (38 mg/L) were somewhat greater than the observed lake mean. Mean magnesium concentrations in Rice Lake were lower than either estimated concentrations* (9 mg/L) or the raw water samples collected from the Bissett water treatment plant (7 mg/L; Ibid). The inference was that excess calcium and magnesium cations from the dewatering process had raised the mean total alkalinity of Rice Lake and thereby effected a slight elevation of mean pH levels.

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* Calculated by the same procedure used in previously estimating conductivity.

Gold had been recovered in the San Antonio mill by the amalgamation and cyanidation process (Anon., 1956). Thus it was imperative that residual concentrations of cyanide in Rice Lake were quantified. However, cyanide was not detected in water samples collected during the pre-reactivation sampling period.

Cadmium also was not detected during the pre-reactivation sampling period. This was consistent with conditions extant in Rice Lake prior to mineshaft dewatering (Moche, 1980). Additionally, complete lake dilution (dilution factor = 0.0093) of mean shaft water cadmium concentrations (0.033 mg/L; Ibid) presented a theoretical loading rate of 3.1×10^{-4} mg/L; a concentration well below the current 0.005 mg/L limit of detection. Cadmium typically is not detected in Manitoba surface waters unless gross loadings of metals has occurred (Water Standards and Studies Section, unpublished data).

Arsenic was not detected in any of the water samples. Although not listed as present in the San Antonio ore body, arsenopyrite was an important goldbearer in the Jeep and Ogama Rockland mining operations (Brinco Mining Limited, 1981). Ore from these operations had been trucked to the San Antonio mill for processing (Davies et al, 1962). Shaft dewatering was not considered a significant potential source of arsenic loading as, like cadmium, mean shaftwater concentrations were too minimal (0.004 mg/L; Moche, 1980) to be detected following complete lake dilution.

Relatively low concentrations of copper (mean = 0.042 mg/L) were detected in the water column. These concentrations were not significantly different from levels measured in Rice Lake prior to shaft dewatering (mean = 0.025 mg/L; Ibid) or those in the shaft water itself (mean = 0.045 mg/L; Ibid). Because pristine waters in similar geophysical settings such as Gods Lake (mean = 0.025 mg/L; Beck, 1982) and Sherlett Lake (mean = 0.022 mg/L; Beck, 1983) possess slightly lesser copper concentrations, it was uncertain whether anthropogenic loading was implicated or whether the observed concentrations were natural background.

Lead was detected in eight of the fifteen samples collected. Seven of these showed trace levels at the 0.005 mg/L limit of detection while one sample registered a 0.010 mg/L concentration of lead. Typically, lead is detected infrequently in surface waters. The inference was that anthropogenic loading was implicated. Such loading could have been a product of mine shaft dewatering. Shaft water mean concentrations of lead (0.43 mg/L; Moche, 1981) produced a theoretical loading to Rice Lake of 0.004 mg/L following complete dilution. The theoretical loading, in conjunction with background levels*, could have produced the detections observed.

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* Background concentrations reported by Moche (1981) are suspect in this instance because the November 12, 1980 value of 0.10 mg/L is suggestive of either a contaminated sample container or a gross analytical error.

Nickel was detected at fourteen of the fifteen stations. Nickel concentrations alone among the metals appeared to indicate the greatest relative degree of anthropogenic impact. Although the greatest concentration detected (0.02 mg/L at Station 1) was not particularly excessive, nickel rarely exceeds the 0.005 mg/L level in surface waters located within similar geophysical settings (Beck 1982 and 1983; Brown et al, 1980). The shaft dewatering process was again implicated. However, because mean shaft water concentrations (0.14 mg/L; Moche, 1981) were alone insufficient to theoretically yeild detectable nickel concentrations following complete lake dilution, an other, as yet undetermined source(s) of nickel loading was possible.

Minimal concentrations of zinc were detected in all water samples. However, the mean (0.02 mg/L) was considered to lie within the range common to unimpacted sheild lakes such as Gods Lake (mean = 0.026 mg/L; Beck, 1982), Sherlett Lake (mean = 0.01 mg/L; Beck, 1983) and Wekusko Lake (mean = 0.021; Brown et al, 1980). The mine dewatering process appeared to be an insignificant source of zinc loading as shaft water mean concentrations (0.082 mg/L; Moche, 1981) were theoretically insufficient to yeild detectable levels following complete lake dilution.

Mercury was detected in four of the fifteen water samples collected. Because detection of mercury is atypical of surface water quality monitoring within the province (Water Standards and Studies Section unpublished data), contamination of the Kemmerer water sampler was suspected. (Contamination of sample containers was considered

unlikely because the samples were collected in triplicate, controls were used and elaborate clean up procedures were employed). Subsequent experimentation at the W. M. Ward Technical Services Laboratory confirmed that doubly distilled water readily extracted detectable concentrations of mercury from the Kemmerer sampler. (A. Jakilazek, pers. comm.). Thus, all positive values were considered suspect and reliance was placed only on the "undetected" data. Hence, pre-reactivation concentrations of mercury were considered to be less than the 0.02 µg/L limit of detection.

(b) Post-Reactivation (Ice Cover Conditions)

The March 23, 1982 survey was initiated as a result of the January 27, 1982 failure of the dykes retaining the tailings. At that time, the volume of Rice Lake was unknown and caution dictated that a survey be undertaken in order to assess the impacts of the accidental discharge. Retrospectively, it was unlikely that the tailings discharge had much potential for deleterious impact. Firstly, tailings were discharged to the ice surface and deleterious materials would have been released gradually to the water column as the ice melted. Thus the shock of impact associated with a sudden, massive influx of tailings would have been considerably lessened. Secondly, the volume of tailings discharged was relatively minor in relation to the total volume of Rice Lake. The potential dilution factor following complete mixing of the tailings approximated 68 m^3/9.75 X $10^6/m^3$ or 6.97 X 10^{-6}. Concentrations of deleterious substances would have had to have been very large in order to be detectable following complete lake mixing.

Prior to discussion of results, it must be noted that ice formation can influence water chemistry. Molecules other than water are generally excluded from the lattice during ice formation. The net effect of ice formation is therefore an increase in the concentration of ions within the remaining liquid portion of the water column. Because ice thickness measured approximately 0.6 m during the March 23, 1982 survey, the empirical factor of concentration approximated 2.55 m (2.55 m - 0.60 m), or 1.28. Water column chemical concentrations greatly at variance with the May, 1981 concentration multiplied by the empirical factor (i.e. 1.28) implied that; a) the May 1981 concentrations were not at equilibrium; or b) that the concentrating effect of ice formation triggered some form of reaction; or

c) that anthropogenic effects were implicated.

Conductivity, as a function of total ion activity, was considered a prime candidate for influence by ice formation. However, the ice cover mean conductivity (301.3 uS/cm) was significantly lower than that of the pre-reactivation survey and showed considerably greater variation (δ = 571.0). The inference was that pre-reactivation conductivity had been in a state of flux and had not achieved equilibrium by the time of the March survey. Anthropogenic causes of the observed March variation were not directly implicated. Optimum curve fitting regressions calculated on estimated distances from either the mine dewatering outfall site or the site of the tailings spill versus conductivities failed to yeild a significant (maximum r^2 = 0.23) correlation.

TABLE 2: SUMMARY OF POST-REACTIVATION (ICE COVER) RICE LAKE WATER PHYSICOCHEMISTRY.

PARAMETERS	STATISTIC:	MEAN	STANDARD DEVIATION	GREATEST VALUE	SMALLEST VALUE	NUMBER OF SAMPLES
Field Temperature	(°C)	0.96	0.72	2.5	0.0	12
Conductivity	(uS/cm)	301.	57.0	356.	179.	15
Residue Non Filt.	(mg/L)	7.80	1.42	10.	<5.	15
pH (log.$\bar{X}$)	(pH Units)	7.14	--	7.30	6.80	15
Alkalinity Total ($CaCO_3$)	(mg/L)	69.80	6.87	82.0	54.0	15
Alkalinity (HCO_3)	(mg/L)	85.15	8.37	100.0	65.9	15
Alkalinity (CO_3)	(mg/L)	0.0	0.0	0 0	0.0	15
Alkalinity (OH)	(mg/L)	0.0	0.0	0.0	0.0	15
Dissolved Oxygen	(mg/L)	7.65	3.28	11.9	0.1	15
T.O.C.	(mg/L)	28.77	3.29	36.0	25.0	15
T.I.C.	(mg/L)	18.00	2.24	21.5	13.5	15
Turbidity	(NTu)	1.13	0.30	2.0	0.9	15
Kjeldhal Total-N	(mg/L)	0.88	0.19	1.2	0.7	15
NO_3-NO_2-N (Diss.)	(mg/L)	0.12	0.05	0.16	0.03	15
P-Total	(mg/L)	0.026	0.018	0.08	0.01	15
Cyanide (Total)	(mg/L)	<0.02	0.0	<0.02	<0.02	15
As (Total)	(mg/L)	0.002	0.0015	0.007	0.001	15
Cd (Ext.)	(mg/L)	<0.005	0.0	<0.005	<0.005	15
Cu (Ext.)	(mg/L)	0.031	0.0247	0.106	<0.005	15
Pb (Ext.)	(mg/L)	<0.005	0.0	<0.005	<0.005	15
Ni (Ext.)	(mg/L)	<0.005	0.0	<0.005	<0.005	15
Zn (Ext. Dir.)	(mg/L)	0.042	0.0329	0.151	0.011	15
Ca (Ext.)	(mg/L)	32.97	6.09	38.4	21.0	15
Mg (Ext.)	(mg/L)	9.181	1.894	10.90	5.27	15
Hg (Total)	(µg/L)	<0.02	0.0	<0.02	<0.02	15

Non filterable residues increased significantly over the pre-reactivation sampling period. Though statistically significant, the increase (May, 1981 mean = <5.0 mg/L; March, 1982 mean = 7.80 mg/L) was insufficient to effect a significantly elevated turbidity. This is contrary to the situation usually encountered. Typically, turbidity and non filterable residues are greater during open water periods when wind and wave action retain particles in suspension in the water column. The observed increase in non filterable residues during this ice cover period was attributed to incomplete decomposition of rooted macrophytes which proliferated during the open water season. Neutrally buoyant particles of plants were observed at many stations and both the significant increase in total organic carbons and the significant decrease in dissolved oxygen saturation levels (discussed later) were also supportive of this view.

Nutrients were expected to increase due to both physical concentration by ice formation and to bacterial releases from decomposed vegetation (i.e. Hutchinson's (1957) classic nitrogen cycle). Significant increases in both the dissolved nitrogens and total Kjeldhal nitrogen were attributable to these influences*.

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* For example, applying the ice concentration factor to mean TKN concentrations for May 1980, plus the observed increases in the nitrite and nitrate forms of nitrogen [(1.28 x 0.6 mg/L) + (0.12 mg/L - 0.04 mg/L) = 0.85 mgN/L] yeilds a close approximation of observed mean TKN.

The mean total phosphorus concentration (0.026 mg/L) was apparently uninfluenced by these factors. However, significant differences may have been masked by the large analytical error (approximately 50% at the 0.02 mg/L level; Sorba et al, 1980) associated with lower limit phosphorus analyses.

Mean dissolved oxygen concentrations (7.65 mg/L) demonstrated a significantly reduced mean level of saturation (54%) when compared to May, 1980 (95%). This response is typical of shallow lakes during ice cover conditions and was attributable to several factors. The physical presence of the snow and ice cover prevented direct atmospheric dissolution and hindered the transmission of sunlight. Sunlight was virtually eliminated causing the death of much of the macrophytic growth amd a decrease in phytoplankton photosynthesis. Dissolved oxygen concentrations were also actively reduced by bacterial decomposition of the macrophytes and other organic matter.

Normally, elevated alkalinities are associated with higher pH values and lower alkalinities with lower pH values. However, significant increases in both total and bicarbonate alkalinities were, in this instance, associated with a significant decrease in pH values. One possible explanation involves the dissociation of carbonic acid (Hutchinson, 1957). It was thought that the decomposition of vegetation (previously discussed) had produced large quantities of carbon dioxide. Upon dissolution in the water column, carbonic acid is formed from carbon dioxide. Dissociation of large quantities of carbonic acid, a ready occurrence, would be

"signalled" by elevated bicarbonate ion concentrations, a decrease in pH and little change in calcium and magnesium concentrations (Ibid). Excepting a nominal (though significant) increase in magnesium concentrations, the "signals" of carbonic acid dissociation were observed. Changes in alkalinities and pH values during ice cover conditions were therefore attributed to natural organic decomposition and not to anthropogenic sources.

The San Antonio Joint Venture, unlike the previous operation, opted for gold recovery solely by the flotation process. Thus it was not surprising that cyanide was again not detected in the water column of Rice Lake.

Cadmium was not detected during the March, 1982 survey. This was consistent with the assessment of the May, 1981 survey results.

Unlike the previous survey, arsenic was detected at all stations sampled during the March, 1982 survey. Levels ranged from 0.001 mg/L to 0.007 mg/L with a mean of 0.002 mg/L. This represented a significant increase from the open water concentrations. Simple concentration due to the 1.28 ice formation factor (previously derived) could not account for the increase from background (<0.001 mg/L). Anthropogenic sources also were not directly implicated as metals more likely to indicate loading (eg. copper or lead) showed no evidence of significantly elevated concentrations. Decomposing macrophyte tissues did not appear to be a likely source as vascular plant tissues cannot tolerate a large body burden without suffering growth retardation (Leibig, 1966). The only remaining potential source considered probable was the sediments, from which arsenic may have been released anaerobically.

Mean copper concentrations (0.031 mg/L) were lower than during the May, 1981 survey. However, the difference was not significant at $t\alpha = 0.05$. This tended to support the possibility that the mine shaft dewatering process had loaded significant amounts of copper to Rice Lake. Further indications were that the May, 1981 copper concentrations had been in a state of flux and that equilibrium had not been achieved at the time of sample collection.

Lead was not detected at any stations sampled during the March, 1982 survey. Although this represented a decrease from the May, 1981 water column lead concentrations, the difference once again was not significantly ($t\alpha = 0.05$) different. This tended to support previous discussion wherein anthropogenic loading via the mine shaft dewatering process was inferred. The data also suggested that <0.005 mg/L represented the normal background lead concentration in the water column.

Nickel, like lead, was not detected at any of the stations sampled during the March, 1982 survey. Failure to detect nickel supported the earlier view that previous detections of nickel in the water column were indicative of an abnormal situation; inferring anthropogenic loading.

Mean zinc concentrations (0.042 mg/L) appeared significantly elevated over those of May, 1981 (0.021 mg/L). Like arsenic, the increase could not be attributed fully to the 1.28 ice formation factor, decomposition of organic matter or anthropogenic sources. Although anaerobic release from the sediments could not be entirely

ruled out, the cause of the apparent increase may have been analytical. Previous analyses were limited to extractable zinc whereas analyses of the March, 1982 samples involved measurement of total zinc concentrations.

Due to difficulties encountered during the May, 1981 survey regarding mercury contamination of the Kemmerer sampler, the March, 1982 samples were collected by hand submersion of the sample bottles. Mercury was not detected at any of the stations sampled in this manner.

c) Post-reactivation (Open Water Conditions)

It was intended that the June, 1982 sampling period provide data directly comparable to that of the pre-reactivation sampling period. The purpose was to identify of water quality changes induced by reactivation of the mine and mill operation.

The mean specific conductance (205 µS/cm) showed a further significant decline from previous sampling periods. This was viewed as confirming that conductivities encountered during May, 1981 had been significantly elevated over background by discharges of mine shaft water. The relatively small coefficient of variation encountered during the June, 1982 survey ($\delta/x = 0.014$) was interpreted as indicative of a nearing state of equilibrium. The inferrence was that lake conductivities were likely to remain somewhat elevated over the pre shaft water discharge levels (151 µS/cm) reported by Moche (1981).

TABLE 3: SUMMARY OF POST-REACTIVATION (OPEN WATER) RICE LAKE WATER PHYSICOCHEMISTRY.

PARAMETERS	STATISTIC:	MEAN	STANDARD DEVIATION	GREATEST VALUE	SMALLEST VALUE	NUMBER OF SAMPLES
Field Temperature	(°C)	16.5	.74	18.0	15.0	15
Conductivity	(uS/cm)	205.	2.94	209.	198.	15
Residue Non Filt.	(mg/L)	<5.	0.0	<5.	<5.	15
pH (log.$\bar{x}$)	(pH Units)	7.69	--	7.80	7.55	15
Alkalinity Total ($CaCO_3$)	(mg/L)	47.6	1.18	49.0	45.0	15
Alkalinity (HCO_3	(mg/L)	58.1	1.45	59.8	55.0	15
Alkalinity (CO_3)	(mg/L)	0.0	0.0	0.0	0.0	9
Alkalinity (OH)	(mg/L)	0.0	0.0	0.0	0.0	8
Dissolved Oxygen	(mg/L)	7.6	0.2	7.9	7.3	15
T.O.C.	(mg/L)	19.9	0.4	20.0	19.5	15
T.I.C.	(mg/L)	9.5	0.63	10.0	8.0	15
Turbidity	(NTu)	0.9	0.05	1.0	0.8	15
Kjeldhal Total-N	(mg/L)	0.6	0.35	0.7	0.6	15
NO_3-NO_2-N (Diss.)	(mg/L)	0.02	0.007	0.03	0.01	15
P-Total	(mg/L)	0.025	0.0038	0.032	0.018	15
Cyanide (Total)	(mg/L)	<0.02	0.0	<0.02	<0.02	15
As (Total)	(mg/L)	0.001	0.0008	0.004	<0.001	15
Cd (Ext.)	(mg/L)	<0.005	0.0	<0.005	<0.005	15
Cu (Ext.)	(mg/L)	0.02	0.013	0.05	0.01	15
Pb (Ext.)	(mg/L)	<0.005	0.0	<0.005	<0.005	15
Ni (Ext.)	(mg/L)	<0.005	0.0	<0.005	<0.005	15
Zn (Ext. Dir.)	(mg/L)	<0.01	0.0	<0.01	<0.01	15
Ca (Ext.)	(mg/L)	22.9	0.70	24.1	21.4	15
Mg (Ext.)	(mg/L)	7.48	0.290	8.06	6.91	15
Hg (Total)	(µg/L)	<0.02	0.0	<0.02	<0.02	13

TABLE 4: COMPARISON AND RANK OF RICE LAKE WATER PHYSICOCHEMISTRY BY SAMPLING PERIOD.

PARAMETER	STATISTIC: SIGNIFICANT DIFFERENCES PER ANALYSIS OF VARIANCE F	d.f.	Fo.95	RANK AND SIMILARITY PER DUNCANS MULTIPLE RANGE TEST (α = 0.05)*
CONDUCTIVITY	65.53	2,42	3.22	ABC
NON FILTERABLE RESIDUE	57.97	2,42	3.22	BAC
pH	61.59	2,42	3.22	ACB
TOTAL ALKALINITY ($CaCO_3$)	126.27	2,41	3.21	BAC
ALKALINITY (HCO_3)	130.86	2,42	3.22	BAC
ALKALINITY (CO_3)	0	2,42	3.22	IDENTICAL
ALKALINITY (OH)	0	2,42	3.22	IDENTICAL
DISSOLVED OXYGEN (% SATURATION)	26.99	2,39	3.24	ACB
TOTAL ORGANIC CARBON	134.97	2,42	3.22	BCA
TOTAL INORGANIC CARBON	142.28	2,42	3.22	BAC
TURBIDITY	2.08	2,42	3.22	BAC
TOTAL KJELDHAL NITROGEN	35.08	2,42	3.22	BAC
DISSOLVED NITRATE-NITRITE	32.32	2,42	3.22	BAC
TOTAL PHOSPHORUS	0.56	2,42	3.22	ABC
CYANIDE	0	2,42	3.22	IDENTICAL
ARSENIC	5.44	2,42	3.22	BCA
CADMIUM	0	2,42	3.22	IDENTICAL
COPPER	5.33	2,42	3.22	ABC
LEAD	1.00	2,42	3.22	ABC
NICKEL	17.50	2,42	3.22	ABC
ZINC	9.36	2,42	3.22	BAC
CALCIUM	31.40	2,42	3.22	BAC
MAGNESIUM	54.21	2,42	3.22	BCA
MERCURY	0	2,38	3.25	IDENTICAL

* SAMPLING PERIODS ARE RANKED FROM HIGHEST TO LOWEST VALUE WHERE A = MAY 1981, B = MARCH 1982 AND C = JUNE 1982. STATISTICALLY SIMILAR SAMPLING PERIODS ARE UNDERSCORED.

Mean non filterable residues (<5.0 mg/L) showed a significant decline from March, 1982 levels. This was viewed as further evidence that elevated non filterable residues during the ice cover sampling period had been induced by partially decomposed macrophytic particles in the water column. However, like the previous sampling period, the June, 1982 change in non filterable residues was insufficient to significantly affect mean turbidities (0.9 NTu). These remained virtually indistinquishable from the values of previous sampling periods.

It will be recalled that previous nutrient levels had not been detectably affected by anthropogenic sources. This continued to be the case. Both the mean total Kjeldhal nitrogen (0.6 mg/L) and the dissolved forms of nitrogen (0.02 mg/L) were not significantly different from the May, 1981 concentrations. Additionally, mean total phosphorus (0.025 mg/L) showed no significant change from previous sampling periods.

Mean dissolved oxygen concentrations (7.6 mg/L) were somewhat lower than anticipated for open water conditions. However, the mean level of saturation (78%) was not significantly different from that of May, 1981.

The pH values ranged from 7.55 to 7.80 units with a logarithmic mean of 7.69 units. This was more comparable to the May, 1981 values (7.87 units) than to those of the March, 1982 (7.10 units) ice cover sampling period. Mean total and bicarbonate alkalinities (47.6 and 58.1 mg/L; respectively) were significantly lower than during either of the two previous sampling periods. This

tended to corroborate earlier discussion which linked midwinter declines in pH values to the dissociation of carbonic acid in the manner described by Hutchinson (1957). The significant decrease in mean calcium concentrations (22.9 mg/L) supported previous assertions that mine shaft water discharges had resulted in calcium loadings. Previous assertions that magnesium loading had also occurred were not supported as mean concentrations (7.48 mg/L) were marginally (though significant) greater than during May, 1981.

Again, cyanide was not detected at any of the stations sampled.

Cadmium was not detected during the June, 1982 sampling period. This continued to be consistent with results from the preceeding sampling periods.

Trace levels of arsenic (maximum, 0.004 mg/L) were detected at three of the fifteen stations sampled. The mean (<0.001 mg/L) was significantly lower than the ice cover period but was not significantly different from the May, 1981 sampling period. This supported the earlier suggestion that slight amounts of arsenic had entered the water column of Rice Lake following reactivation of the mine and mill. However, inferences regarding the source(s) of arsenic could not be drawn solely from a comparison of water chemistry data.

Mean copper concentrations (0.02 mg/L) continued to exhibit a negative trend. Concentrations were lower than during both the May, 1981 sampling period (significant) and that of March, 1982 (not significant). This tended to lend support to the earlier suggestion that the mine shaft dewatering process had loaded significant amounts of copper to the water column of Rice Lake.

Consistent with the ice-cover sampling period, lead was not detected during the June, 1982 survey. This supported the earlier inference that <0.005 mg/L represented the normal background lead concentration in the water column.

Nickel again was not detected at any of the stations sampled. This tended to confirm earlier suggestions that the nickel detected during the May, 1981 sampling period was indicative of anthropogenic loading. The inference that normal background nickel concentrations were less than the 0.005 mg/L limit of detection was also supported.

Zinc was not detected in any of the samples analyzed. This represented a significant reduction from both of the previous sampling periods. It suggested that previous assumptions may have been invalid and that the mine shaft dewatering process may have indeed effected a slight loading of zinc to the water column of Rice Lake. The suggestion, however, was unsupported by other data.

Water column mercury concentrations continued to be below the 0.02 µg/L limit of detection.

d) Assessment of Impacts

Prior to summarizing the assessment of impacts, it was necessary to briefly review the potential sources of anthropogenic impact upon Rice Lake. Chronologically, previous mining activity was the first potential anthropogenic source. The present study could address the issue only under the broadest of terms due to lack of a background (i.e. prior to 1932) data base. Comparisons to other water

bodies provided only a generalized view of true background. The mine shaft dewatering process was the second potential anthropogenic source. The third potential source of anthropogenic impact was, of course, operation of the reactivated mine and mill. This was the potential source which the present study had been designed to address.

Previous mining activity appeared to have had little, if any, lasting impact upon the water column physicochemistry of Rice Lake. Only specific conductance appeared somewhat elevated beyond what might have been expected for lakes in a similar geophysical setting. Other parameters did not appear appreciably affected under the very generalized terms of comparison used. This was consistent with effects of the reactivated operation.

Mine shaft dewatering effected the most significant changes to Rice Lake water physicochemistry noted during the present study. Elevated levels of conductivity, pH, alkalinity, buffering capacity, copper and lead were all attributed to the dewatering process. Water column nickel concentrations also increased significantly following mine shaft dewatering, but a definite link could not be established. However, with the possible exception of conductivity, all the affected parameters achieved equilibrium at what apparently had been background levels. This equilibrium was achieved within eighteen months of completion of the mineshaft dewatering process.

A slight increase in water column arsenic concentrations appeared to be the sole effect of operation of the reactivated mine and mill. However, other more subtle impacts, may have been masked by the effects of the dewatering process.

Sediments and Tailings

Metal and cyanide concentrations in Rice Lake sediments and San Antonio Mine tailings are detailed in Appendix 4. This data is summarized in Table 5. Regressions of sediment metal concentrations versus distance from the tailings area are presented in Table 6. This data was used to assess previous mining activity and to provide background for the reactivated operation.

Mercury analyses were reported for both "wet" sediment samples and on samples dried to a constant weight. The purpose of "wet" analyses was to eliminate net loss of mercury to volatilization during the drying process.

Mean mercury ("dry") concentrations in lake sediments (1.42 µg/g) were significantly greater ($t = 7.96$) than mean tailings concentrations (0.20 µg/g). Optimum curve fitting regressions calculated on mercury concentrations versus distance from the tailings showed a significant (maximum $r^2 = 0.71$) negative association between concentrations and distance. The optimal association took the form of a power curve ($y = a\ x^b$) where µg/g Hg $= 0.06\ m^{0.48}$ (Figure 4). Because of the concentration difference and the negative

association, the obvious inference was that mercury had been actively purged from the tailings and transported away from the tailings area. Post depositional (therefore, natural) mechanisms for purging of the tailings are pH dependent (Schmidt et al, 1981) and would have required either acidification or strong alkalization of the tailings area and Rice Lake. The case for a post depositional mechanism was not substantiated by either alkalization or acidification of the Rice Lake water column. The implication was that mercury purging from the tailings was pre-depositional and, due to previous milling practices, likely an effect of the cyanidation process. The implied mechanism involved dropping of purged tailings near the outfall while effluents laden with the cyanide - mercury complexes were dispersed. The probability of complex destabilization (hence, sediment deposition) due to biodegradation and photodecomposition would be a function of residence time (Ibid); itself a function of distance from the outfall. A similar mechanism resulting in loading of other metals to a retention pond has been harnessed for treatment of gold milling effluents (Ibid). Thus it seemed reasonable to conclude that one long term effect of the previous San Antonio operation had been the loading of mercury to the sediments of Rice Lake.

TABLE 5: SUMMARY OF METAL AND CYANIDE CONCENTRATIONS IN RICE LAKE SEDIMENTS AND SAN ANTONIO MINE TAILINGS

SUBSTRATE	METAL	STATISTIC* $\bar{x}$	δ	Hi	Lo	N
SEDIMENTS	Mercury (µg/g,wet)	0.28	0.12	0.59	0.08	15
	Mercury (µg/g,dry)	1.42	0.49	2.08	0.11	15
	Arsenic (µg/g)	32.71	9.20	45.5	14.5	15
	Cadmium (µg/g)	1.93	0.26	2.0	<1.0	15
	Copper (µg/g)	610.4	229.88	1020.	36.	15
	Lead (µg/g)	33.7	6.4	45.	20.	15
	Nickel (µg/g)	53.5	10.5	75.	31.	15
	Zinc (µg/g)	202.5	43.11	252.	70.	15
	Iron (%)	3.74	0.67	5.31	2.73	15
	Manganese (µg/g)	382.7	125.61	650.	210.	15
	Cyanide (µg/g)	<1.0	0	<1.0	<1.0	3
TAILINGS	Mercury (µg/g,wet)	0.16	0.11	0.28	0.08	3
	Mercury (µg/g,dry)	0.20	0.15	0.37	0.10	3
	Arsenic (µg/g)	16.3	6.81	24.	11.	3
	Cadmium (µg/g)	1.7	0.58	2.	<1.	3
	Copper (µg/g)	64.0	6.93	72.	60	3
	Lead (µg/g)	28.3	20.21	50.	10.	3
	Nickel (µg/g)	56.	19.00	75.	37.	3
	Zinc (µg/g)	182.0	124.43	322.	84.	3
	Iron (%)	46.9	2.80	49.7	44.1	3
	Manganese (µg/g)	726.7	40.41	770.	690.	3
	Cyanide (µg/g)	5.0	5.20	11.	2.	3

*Note: $\bar{x}$ = Mean
δ = Standard Deviation
Hi = Greatest Value
Lo = Smallest Value
N = Number of Samples

TABLE 6: REGRESSIONS OF SEDIMENT METAL CONCENTRATIONS (µg/g)* VERSUS DISTANCE (m) FROM THE MINE SITE

METAL	WHERE µg/g = $a\ m^b$ a	b	r^2	Ho:β = 0 (t)***
Mercury (wet)	0.09	0.17	0.23	ACCEPT (1.97)
Mercury (dry)	0.06	0.48	0.71	REJECT (5.64)
Arsenic	18.22	0.08	0.10	ACCEPT (1.20)
Cadmium	1.89	0.002	<0.01	ACCEPT (0.36)
Copper	11.35	0.59	0.88	REJECT (9.76)
Lead	16.12	0.11	0.45	REJECT (3.26)
Nickel	25.76	0.11	0.41	REJECT (3.01)
Zinc	54.35	0.20	0.65	REJECT (4.91)
Manganese	960.05	-0.15	0.36	REJECT (2.70)
Iron**	5.53	-0.06	0.20	ACCEPT (1.80)

* Regression for Iron is based on % instead of µg/g.

** A better fit (r^2 = 0.46) was obtained for the linear equation where % Fe = m (-0.001) + 4.70. The Ho:β = 0 for this equation was rejected (t = 3.33).

*** Critical t at α = 0.05 was 2.16 (d.f. = 13).

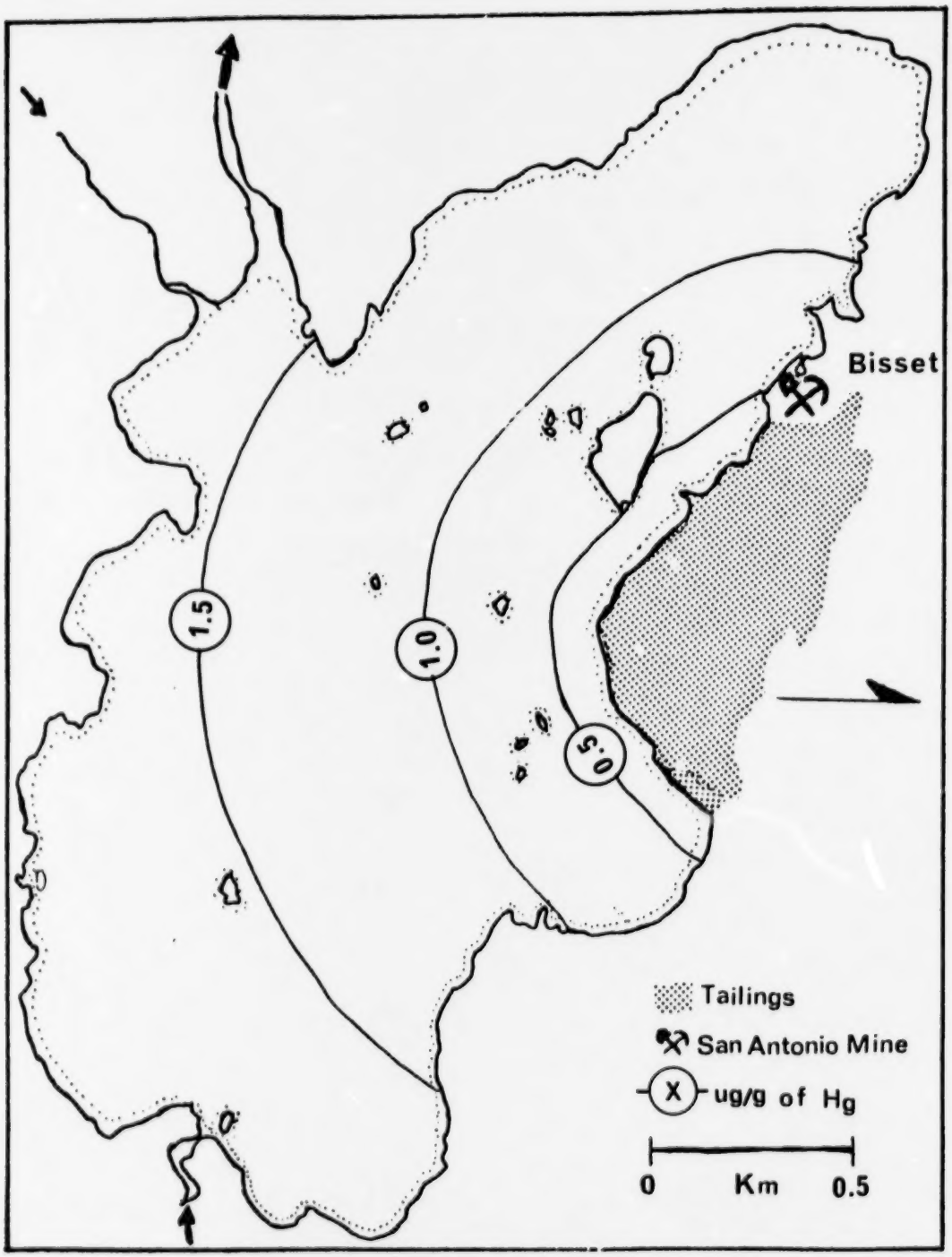

Figure 4: Isopleths of mercury (µg/g) in Rice Lake sediments.

Mean arsenic concentrations in lake sediments (32.71 µg/g) were approximately twice that of the tailings area (16.3 µg/g). This difference was significant ($t = 3.57$) at $t\alpha = 0.05$. Unlike mercury, a regression showing a significant association between arsenic concentrations and distance from the tailings area could not be developed from the data. It was concluded that although the previous San Antonio operation may have effected a reduction of arsenic levels in the tailings (compared to background levels in ores), a significant loading to the sediments of Rice Lake was not implicated. This conclusion was consistent with water column physicochemistry results of the pre-reactivational sampling period.

Mean cadmium concentrations in lake sediments (1.93 µg/g) and the tailings (1.7 µg/g) were statistically indistinquishable ($t = 0.67$). Thus, it was not surprising that a significant correlation linking cadmium concentrations to tailings proximity could not be established. It appeared that the cadmium concentrations encountered were normal background for the area and that levels had not been influenced by previous mining activity.

The status of copper was similar to that of mercury. Mean lake sediment concentrations (610.4 µg/g) were approximately one order of magnitude greater than mean tailings concentrations (64.0 µg/g). Additionally, copper concentrations showed a significant ($r^2 = 0.88$), negative association with distance from the tailings area. Once again, the relationship took on the form of a power curve whereby µg/g Cu = $11.35\ m^{0.59}$ (Figure 5). Like mercury, the data was interpreted as an indication that the previous San Antonio operation had effected a significant loading of copper to the sediments of Rice Lake.

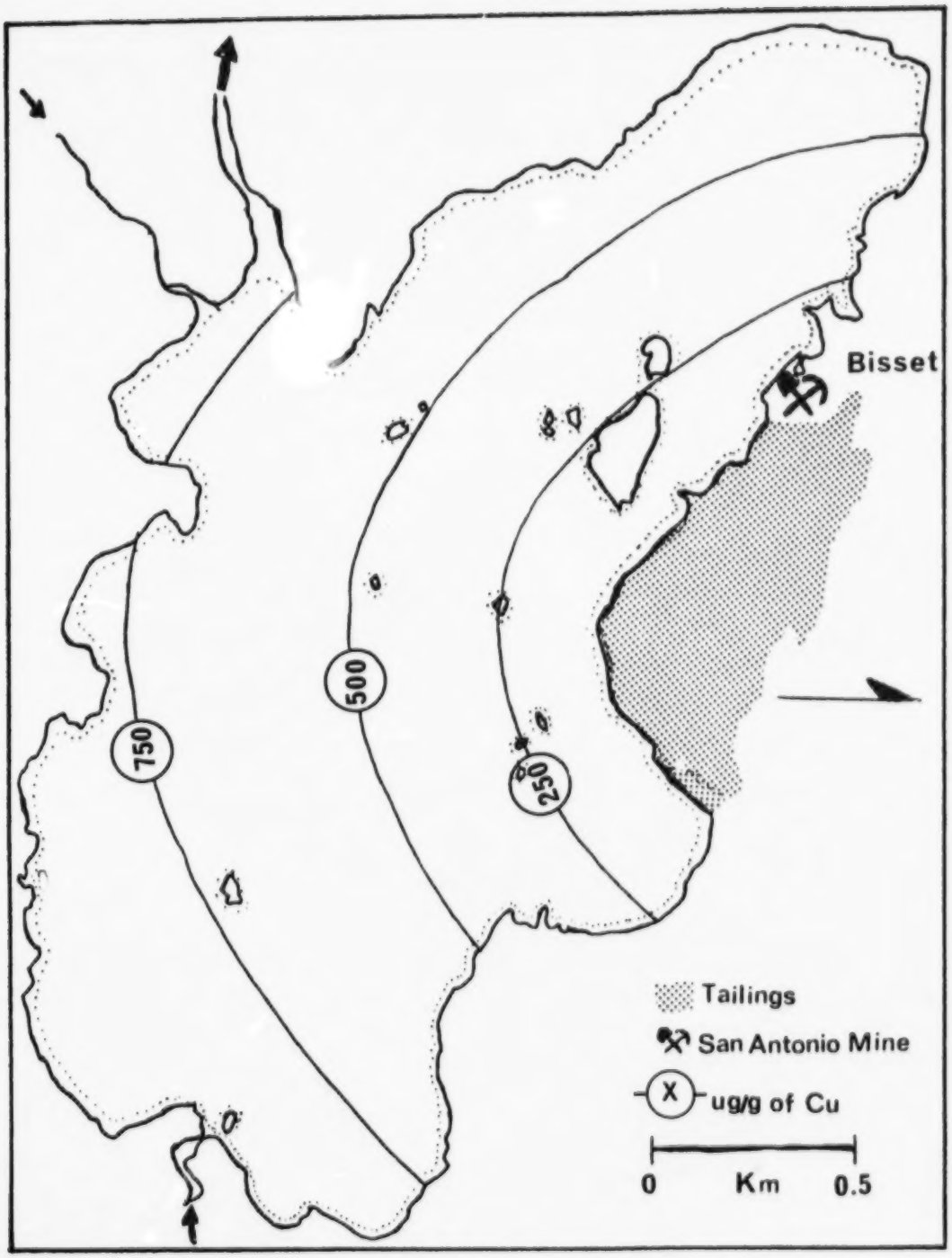

Figure 5: Isopleths of copper (µg/g) in Rice Lake sediments.

Mean lead concentrations in lake sediments and tailings were quite similar (33.7 µg/g, and 28.3 µg/g; respectively) and showed no significant difference (t = 0.46). Like copper, lead concentrations showed a significant (r^2 = 0.45) negative association with distance from the tailings area. Also like copper, the inference was that the previous San Antonio operation had effected a significant loading of lead to the sediments of Rice Lake. However, because the empirical evidence (i.e. lake sediment concentrations versus tailings concentrations) did not substantiate the inferential suggestion, the latter was considered misleading and it was concluded that significant loading of lead to the sediments of Rice Lake had not occurred during the previous San Antonio operation.

The status of nickel was most similar to that of lead. Mean nickel concentrations in lake sediments (53.5 µg/g) were nearly identical to tailings concentrations (56.0 µg/g) and a significant (r^2 = 0.41) negative association was noted between sediment concentrations and proximity to the tailings area. Similar to lead results, the nickel data was interpreted as indicative that significant nickel loadings to Rice Lake sediments had not occurred during the previous San Antonio operation.

Zinc closely mirrored the lead and nickel results. Mean sediment and tailings concentrations (202.5 and 182.0 µg/g, respectively) were similar and a significant (r^2 = 0.65) negative association was observed between sediment concentrations and proximity to the tailings area. This, again, was interpreted as indicative that significant zinc loadings to Rice Lake sediments had not occurred during previous operation of the San Antonio mine and mill.

Iron results were considerably different from those of the metals previously discussed. Mean tailings concentrations (46.9%) were more than one order of magnitude greater than mean lake sediment concentrations (3.74%). Additionally, a significant (r^2 = 0.46) positive linear relationship was observed between lake sediment concentrations and proximity to the tailings area. This relationship, whereby %Fe = m(-0.001) + 4.70, clearly implied that iron had been slowly eroded from the tailings area and transported to nearby lake sediments (Figure 6). This development is typical of areas where pyritic ores have been processed and resultant tailings are exposed to the atmosphere.

Although sediment and tailings concentrations were not as distinct, manganese results were comparable to those of iron. Mean tailings concentrations (726.7 µg/g) were approximately double the mean lake sediment concentrations (382.7 µg/g). Unlike iron, the optimal (r^2 = 0.36) curvilinear relationship developed from the data was a power curve, whereby µg/g Mn = 960.05 $m^{-0.15}$. This relationship inferred a positive association between lake sediment concentrations and proximity to the tailings area. Physical erosion and transport of manganese bearing tailings materials were again implicated.

Minor amounts of cyanide (mean = 5.0 µg/g) were detected in the tailings samples. Cyanide was not detected in lake sediments. These results neither supported nor contradicted earlier suggestions that cyanide was implicated in the loading of mercury and copper to Rice Lake sediments.

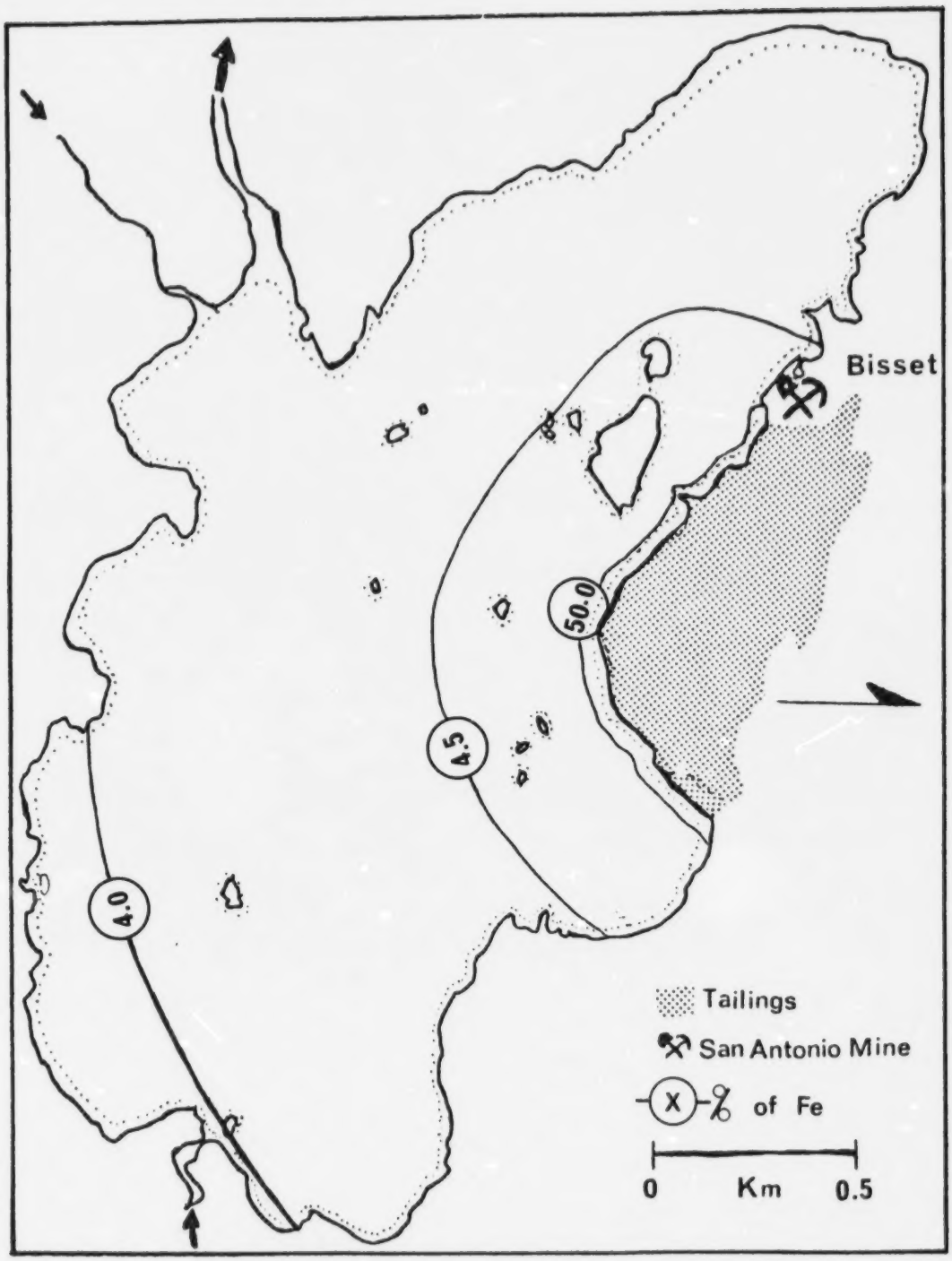

Figure 6: Isopleths of iron (%) in Rice Lake sediments.

Fish

A total of sixty-seven fish were collected and sampled. This included fourty northern pike, eight common sucker and nineteen yellow perch. General physical meristics and muscle tissue metal levels are detailed in Appendix 5. Summaries are presented in Table 7. Comparisions of Rice Lake fish muscle tissue metal concentrations to those of fish from other lakes in a similar geophysical setting are presented in Table 8.

Mean mercury concentrations ranged from 0.54 µg/g in northern pike to 0.30 µg/g in common sucker and 0.37 µg/g in yellow perch. Three perch (16%) and nineteen northern pike (48%) exceeded the 0.50 µg/g level. Only one fish, a northern pike (1.12 µg/g) exceeded the 1.0 µg/g level. Although mean mercury concentrations were not particularly great when compared to concentrations encountered in Winnipeg River fish (Beck 1984(b)), mean concentrations were significantly greater than levels encountered in the same species taken from lakes in similar geophysical settings (Table 8.) Because lakes such as Sherlett, Cold, Snow and Herblet have not been subject to significant mercury loadings (Beck, 1983 and 1984(a), respectively), the inference was that mercury loading to Rice Lake had occurred. This is consistent with assessment of the lake sediment results.

TABLE 7: SUMMARY OF GENERAL PHYSICAL MERISTICS AND MUSCLE TISSUE METAL CONCENTRATIONS IN RICE LAKE FISH

Fish Species (N)	Length (cm)	Weight (g)	Sex M:F	Age (Yrs)	Hg	Muscle Tissue Metal Levels (µg/g) As	Cd	Cu	Pb	Ni	Zn	Statistic*
Northern	44.2	580.5	0.9:1	4.40	0.54	0.03	<0.1	0.91	0.87	<0.61	3.54	$\bar{x}$
Pike	3.48	94.08		0.59	0.19	0.012	0.011	0.54	0.32	0.29	0.50	δ
(N = 40)	53.9	780		6	1.12	0.08	0.2	2.0	2.5	2.0	4.6	Hi
	38.3	400		3	0.28	<0.02	<0.1	0.2	0.5	<0.5	2.4	Lo
Common	46.1	1408.8	0.6:1	8.88	0.30	<0.02	<0.1	0.58	0.60	0.56	3.45	$\bar{x}$
Sucker	3.82	611.12		4.09	0.08	0	0	0.19	0.11	0.18	0.72	δ
(N - 8)	51.2	2810		15	0.42	<0.02	<0.1	0.9	0.7	1.0	4.5	Hi
	40.4	900		5	0.20	<0.02	<0.1	0.4	0.5	<0.5	2.8	Lo
Yellow	29.4	401.6	0.36:1	6.37	0.37	<0.02	<0.1	0.44	0.79	0.58	5.03	$\bar{x}$
Perch	2.76	124.91		1.21	0.11	0	0	0.09	0.26	0.34	0.55	δ
(N = 19)	34.7	620		8	0.59	<0.02	01	0.6	1.2	2.0	5.9	Hi
	24.3	190		5	0.21	<0.02	<0.1	0.3	0.5	<0.5	3.9	Lo

*Note: $\bar{x}$ = mean
δ = standard deviation
Hi = greatest value
Lo = smallest value

TABLE 8: COMPARISON ($t_{\alpha} = 0.05$) OF MUSCLE TISSUE METAL CONCENTRATIONS BETWEEN FISH FROM RICE LAKE AND FISH FROM SNOW LAKE, SHERLETT LAKE, HERBLET LAKE AND COLD LAKE.

LAKE COMPARISON	SPECIES	LAKE[1] WITH SIGNIFICANTLY GREATER CONCENTRATION ($t_{\alpha} = 0.05$)						
		Hg	As	Cd	Cu	Pb	Ni	Zn
Rice vs Snow[2]	Northern Pike	Ri	Sn	--	Ri	Ri	--	Sn
	Common Sucker	Ri	--	--	--	Ri	--	--
Rice vs Sherlett[3]	Northern Pike	Ri	--	--	Ri	Ri	Ri	Sh
	Common Sucker	Ri	--	--	--	--	--	--
	Yellow Perch	Ri	--	--	Ri	Ri	--	--
Rice vs Herblet[2]	Northern Pike	Ri	He	--	Ri	Ri	--	He
	Common Sucker	Ri	He	--	Ri	Ri	--	He
Rice vs Cold[3]	Northern Pike	Ri	--	--	Ri	Ri	Ri	Co
	Common Sucker	Ri	--	--	--	--	--	Co

(1) Lake symbols as follows: Ri = Rice Lake; Sn = Snow Lake; Sh = Sherlett Lake; He = Herblet Lake; Co = Cold Lake; -- = No significant difference.

(2) Means for this lake were extracted from Beck (1984(a)).

(3) Means for this lake were extracted from Beck (1983).

Trace arsenic concentrations were detected (mean = 0.03 µg/g) in northern pike. Arsenic was not detected in either common suckers or yellow perch. Significant differences between Rice Lake fish and those from Sherlett Lake and Cold Lake were not detected. Although Sherlett Lake had not been subject to significant arsenic loading, some loading and transport within the sediment had occurred in Cold Lake (Beck, 1983). Fish from Herblet Lake and northern pike from Snow Lake possessed significantly greater muscle tissue arsenic concentrations than did fish from Rice Lake. Both Snow Lake and Herblet Lake had been subject to significant arsenic loadings (Beck, 1984(a)). These considerations implied that significant arsenic loading to Rice Lake had not occurred. This was consistent with the assessments of both pre-reactivation water chemistry results and the sediment data.

Trace levels of cadmium were detected in one northern pike (0.2 µg/g) and in seven yellow perch (0.1 µg/g). Mean concentrations for each of the species were less than the 0.1 µg/g limit of detection. Cadmium concentrations in Rice Lake fish muscle tissue were thus statistically indistinquishable from concentrations in fish of comparable lakes. Although the frequency of cadmium detection in yellow perch (37%) was greater than anticipated, the data were not considered sufficient to contradict assessments of water column physicochemistry and lake sediment data.

Mean copper concentrations varied from 0.91 µg/g in northern pike to 0.58 µg/g common sucker and 0.44 µg/g in yellow perch. Rice Lake pike possessed significantly greater muscle

tissue copper concentrations than northern pike from each of the four comparable lakes. Common sucker were not as consistent as northern pike since only fish from Herblet Lake were shown to have significantly lower copper concentrations. The northern pike relationships were interpreted as indicative that some loading of copper to Rice Lake had occurred. This was consistent with assessments of both water physicochemistry and the sediment data. The results for common sucker, a species which might be expected to accumulate metals from ingested sediments, were not consistently supportive, possibly because the number of specimens (N = 8) was relatively small.

Mean lead concentrations in fish muscle tissue varied from 0.87 µg/g in northern pike to 0.60 µg/g in common sucker and 0.79 µg/g in yellow perch. Apart from comparisons to common suckers from Sherlett Lake and Cold Lake, all other comparisons consistently showed Rice Lake fish to have significantly greater muscle tissue lead concentrations. This supported earlier discussion wherein water column lead loading via mine shaft dewatering was inferred. The results did not necessarily contradict assessment of sediment data.

Nickel was detected in 18% of the northern pike, 13% of the common suckers and 5% of the yellow perch. Mean muscle tissue concentrations varied from 0.61 µg/g to 0.56 µg/g and 0.58 µg/g, respectively. A consistent significant difference in the lake comparisons was not evident. Though this supported the assessment of sediment data, an inference regarding anthropogenic loading of nickel to the water column prior to the pre-reactivation survey (i.e. May, 1981) could not be established.

Mean zinc concentrations varied from 3.54 µg/g in northern pike to 3.45 µg/g in common sucker and 5.03 µg/g in yellow perch. Rice Lake fish generally possessed significantly lower zinc concentrations than fish from comparable lakes. This supported assessment of the sediment data and also tended to support the view that the mine shaft dewatering process had not effected a significant zinc loading to the water column of Rice Lake.

Benthos

Benthic samples were collected from each of the fifteen random sampling stations. However, due to the large volume of organic detritus in each sample and to very restrictive temporal constraints, analysis of only two samples (stations 1 and 11) was completed. Results are detailed in Table 9.

Benthic organisms present at each of the two stations were indicative of the physical nature of the immediate habitat. Station 1 was typical of most of the lake area. Dense growths of submergent vegetation and a soft organic bottom offered a fertile variety of ecological niches. A total of 211 organisms comprised of 24 taxa were identified from the station 1 sample. Dominant forms included snails (Gastropoda - 57.3 %), midges (Chironomidae - 28.9%) and mayflies (Ephemeroptera - 10.4%). Annelid worms (Oligochaeta) were not encountered.

Station 11 was characterized by a hard bottom of sand-like tailings material. Vegetation was limited to a sparse stand of rushes (Scirpus sp.). The comparatively sterile nature of this site was

TABLE 9: Rice Lake Benthos, May 20, 1981

Taxa	Stn. 1 Number of Organisms	Stn. 1 Relative Abundance Per Group (%)	Stn. 11 Number of Organisms	Stn. 11 Relative Abundance Per Group (%)
Ephemeroptera		10.4%		2.3%
Caenis sp.	22		1	
Trichoptera		0%		4.7%
Limnephilus sp.			1	
Mystacides sp.			1	
Diptera				
Ceratopogonidae	2	0.9%	4	9.3%
Chironomidae		28.9%		30.2%
Natarsia sp.	1			
Procladius sp.	12			
Ablabesmyia sp.	4			
Coelotanypus sp.	6		1	
Chironomus sp.	1			
Kiefferulus sp.	1			
Cryptochironomus sp.	2		1	
Parachironomus sp.			1	
Cladopelma sp.			1	
Microtendipes sp.	2			
Dicrotendipes sp.	2		1	
Glyptotendipes sp.	18		1	
Polypedilum sp.	3			
Endochironomus sp.	3			
Rheotanytarsus sp.			1	
Cladotanytarsus sp.	4			
Tanytarsus sp.			2	
Parotanytarsus sp.	1			
Micropsectra sp.	1		3	
Hydracarina		0%		4.7%
Arrenurus sp.			1	
Unionicola crassipes neartica			1	
Amphipoda		1.4%		0%
Hyalella azteca	3			
Pelecypoda		0.9%		11.6%
Pisidium sp.	2		5	
Gastropoda		57.3%		0%
Helisoma campanulatum	2			
Gyraulus deflectus	1			
Valvata Tricarinata	13			
Probythinella lacustris	83			
Amnicola sp.	22			
Oligochaeta		0%		37.2%
Imm. Tubif. without hair setae			2	
Imm. Tubif. with hair setae			2	
Tubifex sp.			1	
Frayments (unidentified oligochaetes)			11	
TOTAL	211	99.8%	43	100%

evidenced by a lower standing crop. Only 43 benthic organisms were encountered. However, the diversity was relatively greater than station 1 as 21 taxa were represented. Dominent forms included annelid worms (Oligochaeta - 37.2%), midges (Chironomidae - 30.2%) and fingernail clams (Pelecypoda - 11.6%). The high relative diversity encountered at Station 11 suggested that other than the nature of the habitat formed by their physical presence, the pre-reactivation tailings materials had not deleteriously impacted upon the benthic organisms of Rice Lake.

Plankton

A list of phytoplankton species encountered during the present study is presented in Table 10. Estimated surface area (μ^2) occupied by each plankton species is also included. Distribution and densities per sample are presented in Table 11.

A total of 24 taxa were identified. The most numerous order represented at the time of sampling was Chrysophyta. The diatom Asterionella formosa was the most abundant of this group. Other phytoplankton species present at 10 or more sampling stations included the Cyanophyte Anabaena sp., the Chlorophyte Chlorococcus humicola, the Chrysophyte Fragilaria crotonensis and the Pyrrophyte Ceratium hirudinella. The Chrysophytes A. formosa and F. crotonensis occupied the greatest total surface area. The relative species diversity, the preponderance of chrysophytes and the low numbers of cyanophytes were indicative that negligible stress was being imposed upon the primary producers of Rice Lake.

TABLE 10 Species list and estimated surface area (μ^2) of phytoplankton from Rice Lake; June, 1981

TAXA	Surface Area (μ^2)
Cyanophyta	
Anabaena sp.	625
Chroococcus limneticus Lemmermann	400
Oscillatoria sp.	250
Chlorophyta	
Chlorococcum humicola (Naeg.) Rabenhorst	400
Cosmarium sp.	625
Dictyosphaerium pulchellum Wood	1000
Monoraphidium sp.	300
Oocystis sp.	400
Pediastrum Boryanum (Turp.) Meneghini	3500
Scenedesmus quadricauda (Turp.) de Brebisson	650
Sphaerocystis Schroeteri Chodat	2500
Staurastrum sp.	2000
Chrysophyta (Bacillariophyceae)	
Asterionella formosa Hassal	1500
Fragilaria capucina Desmarziers	1250
Fragilaria crotonensis (Ehrenberg) Grunow	12500
Melosira sp.	1200
Navicula sp.	300
Pinnularia sp.	1000
Synedra acus Kuetzing	150
Chrysophyta (Chrysophyceae)	
Dinobryon sp.	750
Synura uvella Ehrenberg	2500
Euglenophyceae	
Euglena sp.	750
Trachelomonas sp.	150
Pyrrophyta	
Ceratium hirundinella (Mueller) Schrank	2500

TABLE 11 Distribution of Phytoplankton at Rice Lake, expressed in numbers per liter; June, 1981. (All values X 10^2)

TAXA	Stn. 1	Stn. 2	Stn. 3	Stn. 4	Stn. 5	Stn. 6	Stn. 7	Stn. 8	Stn. 9	Stn. 10	Stn. 11	Stn. 12	Stn. 13	Stn. 14	Stn. 15
Cyanophyta															
Anabaena sp.	6		8	5	3	2	1	2	1	2	1	6		12	3
Chroococcus limneticus		2													
Oscillatoria sp.										811					
Chlorophyta															
Chlorococcum humicola	2	48	5		9	3	1	4	1	1	1				1
Cosmarium sp.		3			2				1	2		3			
Dictyosphaerium pulchellum										1					
Monoraphidium sp.										3					
Oocystis sp.								1							
Pediastrum Boryanum	1	2			1	1				1					1
Scenedesmus quadricauda										4					
Sphaerocystis Schroeteri							1				1			2	
Staurastrum sp.					1								1		
Chrysophyta (Bacillariophyceae)															
Asterionella formosa	1438	1870	1496	1249	1266	1370	1279	2419	1257	2474	1375	844	1076	1637	1548
Fragilaria capucina		2	2					4		1					
Fragilaria crotonensis	14	41	21	17	16	16	12	34	13	24	21	9	6	24	16
Melosira sp.	2									1	1			1	
Navicula sp.										2					
Pinnularia sp.		1													
Synedra acus										959					
Chrysophyta (Chrysophyceae)															
Dinobryon sp.						3				103		1		1	
Synura uvella												2		2	
Euglenophyceae															
Euglena sp.														1	
Trachelomonas sp.										1					
Pyrrophyta															
Ceratium hirundinella		2	2	3				4	1	1	3	4	2	4	2

CONCLUSIONS

It was concluded that:

1. Historic operation of the San Antonio mine and mill resulted in significant loading of mercury, copper, iron and manganese to the sediments of Rice Lake.

2. Elevated mercury levels in fish muscle tissue were a result of historic loadings of mercury to the sediments of Rice Lake.

3. Historic operation of the San Antonio mine and mill posed no detectable long term negative impacts upon the benthos and phytoplankton of Rice Lake other than the physical change in the nature of habitat offered by submerged tailings.

4. Pre-reactivation discharges of mine shaft water temporarily caused significantly elevated water column levels of conductivity, pH, alkalinity, calcium, lead and copper.

5. Pre-reactivation discharges of mine shaft water effected elevated fish muscle tissue levels of copper and lead.

6. With the possible exception of water column arsenic concentrations, five month operation of the reactivated mine and mill posed no detectable impacts upon the water physicochemistry of Rice Lake.

7. The January 27, 1982 spill of tailings to the ice of Rice Lake posed no detectable impact upon the ice covered water physicochemistry of Rice Lake.

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APPENDIX 1: BASELINE PRE-REACTIVATION WATER PHYSICOCHEMISTRY FOR RICE LAKE, MAY 21, 1981.

PARAMETERS		SAMPLE STATION 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Field Temperature	(°C)	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0	18.0
Conductivity	(uS/cm)	351.	348.	338.	342.	342.	335.	335.	329.	333.	335.	347.	340.	384.	333.	344.
Residue Non Filt.	(mg/L)	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0
pH	(pH Units)	7.80	7.90	7.85	7.7	7.85	7.67	7.87	7.73	7.83	7.83	7.90	7.80	7.85	7.75	7.85
Alkalinity Total ($CaCO_3$)	(mg/L)	51.5	51.0	51.5	51.25	51.0	52.0	5.10	52.0	51.0	51.0	51.0	51.0	232.0	48.0	51.0
Alkalinity (HCO_3	(mg/L)	63.	62.0	63.0	63.0	62.0	63.0	62.0	63.0	62.0	62.0	63.0	62.0	62.0	59.0	62.0
Alkalinity (CO_3)	(mg/L)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0
Alkalinity (OH)	(mg/L)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0 0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dissolved Oxygen	(mg/L)	9.3	8.5	8.9	8.8	8.7	9.0	8.9	9.1	9.4	8.6	8.5	8.9	9.7	9.7	8.9
T.O.C.	(mg/L)	18.0	18.0	18.0	17.5	18.5	17.5	17.5	17.5	17.5	18.5	17.5	18.5	17.5	18.5	18.0
T.I.C.	(mg/L)	12.5	12.0	11.5	11.5	11.5	12.5	11.5	11.5	12.5	11.5	12.0	10.5	12.5	14.0	12.5
Turbidity	(NTu)	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.5	1.0	1.0	1.0	1.0	1.0
Kjeldhal Total-N	(mg/L)	0.6	0.6	0.6	0.5	0.5	0.6	0.6	0.5	0.5	0.6	0.5	0.5	0.5	0.6	0.5
NO_3-NO_2-N (Diss.)	(mg/L)	0.06	<0.01	0.03	0.01	0.14	0.01	0.06	0.05	0.02	0.01	<0.01	0.02	0.02	0.03	0.04
P-Total	(mg/L)	0.03	0.03	0.03	0.08	0.02	0.03	0.03	0.02	0.02	0.04	0.02	0.02	0.02	0.03	0.03
Cyanide (Total)	(mg/L)	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
As (Total)	(mg/L)	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Cd (Ext.)	(mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Cu (Ext.)	(mg/L)	0.065	0.030	0.035	0.095	0.020	0.025	0.025	0.025	0.050	0.040	0.045	0.035	0.065	0.035	0.035
Pb (Ext.)	(mg/L)	0.010	0.005	<0.005	0.005	0.005	0.005	0.005	0.005	<0.005	0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Ni (Ext.)	(mg/L)	0.020	0.010	0.015	0.010	0.010	0.010	0.015	0.010	0.005	0.015	0.005	0.005	0.005	0.010	<0.005
Zn (Ext. Dir.)	(mg/L)	0.02	0.02	0.02	0.03	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Ca (Ext.)	(mg/L)	28.	28.	30.	30.	30.	30.	30.	29.	30.	30.	30.	30.	30.	30.	30.
Mg (Ext.)	(mg/L)	5.	5.	5.	5.	5.	5.	5.	5.	5.	5.	5.	5.	5.	5.	5.
Hg (Total)	(µg/L)	<0.02	<0.02	<0.02	<0.02	<0.02	0.13	<0.02	<0.02	<0.02	<0.02	0.05	0.04	<0.02	0.03	<0.02
Sample Depth	(m)	0.9	0.8	0.8	0.8	2.6	0.5	1.9	0.1	0.8	2.1	0.8	1.3	1.5	1.9	0.1
Total Depth	(m)	2.4	3.0	1.8	2.9	5.2	2.0	2.3	1.7	2.2	3.1	1.0	1.7	2.7	2.3	3.2

APPENDIX 2(a): POST-REACTIVATION WATER PHYSICOCHEMISTRY FOR RICE LAKE DURING ICE COVER CONDITIONS, MARCH 23, 1982.

PARAMETERS		SAMPLE STATION 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Field Temperature	(°C)	1.0	--	1.0	0.0	0.0	1.0	1.0	--	1.0	--	2.0	0.5	2.5	0.5	1.0
Conductivity	µS/cm)	316.	293.	179.	330.	330.	356.	338.	234.	353.	285.	336.	187.	339.	310.	333.
Residue Non Filt.	(mg/L)	7.	8.	7.	8.	7.	9.	7.	10.	7.	6.	8.	10.	9.	<5.	9.
pH	(pH Units)	7.20	7.10	6.85	7.10	7.30	7.30	7.25	6.80	7.25	6.95	7.20	6.70	7.30	7.05	7.20
Alkalinity Total ($CaCO_3$)	(mg/L)	68.0	64.0	62.0	69.0	72.0	76.0	71.0	54.0	78.0	64.0	71.0	82.0	71.0	73.0	72.0
Alkalinity (HCO_3)	(mg/L)	83.0	78.1	75.6	84.2	87.8	92.7	86.6	65.9	95.2	78.1	86.6	100.0	86.6	89.1	87.8
Alkalinity (CO_3)	(mg/L)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Alkalinity (OH)	(mg/L)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0 0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dissolved Oxygen	(mg/L)	8.6	7.6	0.1	7.9	8.0	11.8	9.1	6.7	11.9	8.5	8.6	0.6	9.0	7.9	8.5
T.O.C.	(mg/L)	29.0	30.0	26.5	27.0	23.0	31 0	25.5	36.0	29.0	32.5	28.5	30.0	27.0	31.5	25.0
T.I.C.	(mg/L)	17.0	18.5	19.5	19.0	21.5	16.5	18.0	20.5	20.5	14.0	13.5	18.0	17.0	17.5	19.0
Turbidity	(NTu)	1.0	1.0	1.5	1.0	1.0	1.0	1.0	2.0	0.90	1.0	1.0	1.5	1.0	1.0	1.0
Kjeldhal Total-N	(mg/L)	0.8	0.8	1.2	0.8	0.7	0.9	0.7	1.2	1.0	0.9	0.7	1.2	0.7	0.9	0.7
NO_3-NO_2-N (Diss.)	(mg/L)	0.16	0.14	0.03	0.15	0.16	0.05	0.13	0.13	0.05	0.14	0.15	0.02	0.15	0.15	0.14
P-Total	(mg/L)	0.02	0.02	0.08	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.06	0.02	0.02	0.01
Cyanide (Tot.)	(mg/L)	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
As (Tot.)	(mg/L)	0.003	0.002	0.002	0.002	0.001	0.001	0.001	0.002	0.002	0.001	0.007	0.003	0.001	0.002	0.002
Cd (Tot.)	(mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Cu (Tot.)	(mg/L)	0.014	0.023	0.016	<0.005	0.027	0.032	<0.005	0.038	0.106	0.024	0.018	0.045	0.034	0.052	0.024
Pb (Tot.)	(mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Ni (Tot.)	(mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Zn (Tot.)	(mg/L)	0.044	0.050	0.026	0.011	0.020	0.036	0.021	0.039	0.038	0.022	0.037	0.151	0.038	0.030	0.062
Ca (Ext.)	(mg/L)	35.4	28.4	21.0	34.8	36.0	38.4	36.0	23.7	38.3	33.3	36.2	21.9	37.6	36.9	36.6
Mg (Ext.)	(mg/L)	9.95	7.97	5.27	9.80	10.10	10.90	10.10	6.66	10.70	9.37	10.20	5.50	10.60	10.20	10.40
Hg (Tot.)	(µg/L)	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02

APPENDIX 2(b): ADDITIONAL PARAMETERS RE POST-REACTIVATION WATER PHYSICOCHEMISTRY FOR RICE LAKE DURING ICE COVER CONDITIONS, MARCH 23, 1982.

PARAMETERS		SAMPLE STATION 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Residue Total	(mg/L)	240.	230.	150.	260.	250.	260.	260.	210.	270.	230.	260.	170.	250.	240.	240.
Residue Filt.	(mg/L)	230.	220.	140.	250.	240.	250.	250.	200.	270.	220.	250.	160.	240.	240.	230.
Hardness ($CaCO_3$)	(mg/L)	129.0	104.0	75.0	127.0	132.0	141.0	132.0	86.0	140.0	122.0	133.0	77.0	138.0	134.0	--
Mn (Ext.)	(mg/L)	0.138	0.289	0.289	0.153	0.206	0.050	0.149	0.280	0.055	0.235	0.191	0.320	0.164	0.156	0.220
Fe (Ext.)	(mg/L)	0.311	1.08	0.818	0.367	0.286	0.248	0.306	1.34	0.211	0.475	0.266	0.821	0.293	0.350	0.259
K (Ext.)	(mg/L)	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.
Na (Ext.)	(mg/L)	14.30	13.20	3.29	13.80	15.00	15.90	15.30	7.67	16.00	14.00	15.90	3.42	16.40	13.80	16.10
Colour (True)	(Units)	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.	>50.
NH_3-N (Sol.)	(mg/L)	0.02	0.06	0.17	0.02	<0.02	0.06	<0.02	0.25	0.06	0.07	<0.02	0.19	0.02	0.03	0.02
Cl (Sol.)	(mg/L)	45.	43.	16.	46.	48.	51.	49.	33.	50.	43.	49.	10.	49.	41.	49.
SO_4 (Sol.)	(mg/L)	13.0	12.5	7.2	12.5	13.5	15.0	14.0	11.0	14.5	13.5	14.5	5.0	14.0	12.5	14.0
Sample Depth	(m)	0.8	0.2	0.5	1.3	1.4	0.9	1.4	0.8	1.6	0.21	0.3	0.0	0.0	0.9	0.0
Total Depth	(m)	2.5	1.9	1.2	2.3	2.8	1.6	1.9	1.3	1.8	2.5	1.5	0.6	2.2	1.9	2.2

APPENDIX 3: POST-REACTIVATION WATER PHYSICOCHEMISTRY FOR RICE LAKE DURING OPEN WATER CONDITIONS, JUNE 2, 1982.

PARAMETERS	SAMPLE STATION	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Field Temperature	(°C)	17.0	17.0	17.0	17.0	17.0	16.0	16.0	18.0	16.0	16.0	16.0	17.0	16.0	16.0	15.0
Conductivity	(uS/cm)	207.	202.	207.	207.	209.	207.	204.	198.	204.	207.	207.	208.	204.	202.	207.
Residue Non Filt.	(mg/L)	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.	<5.
pH	(pH Units)	7.70	7.70	7.65	7.75	7.70	7.70	7.70	7.55	7.75	7.60	7.70	7.70	7.70	7.60	7.80
Alkalinity Total ($CaCO_3$)	(mg/L)	47.0	47.0	48.0	49.0	47.0	49.0	47.0	45.0	48.0	48.0	49.0	48.0	47.0	46.0	49.0
Alkalinity (HCO_3	(mg/L)	57.3	57.3	58.6	59.8	57.3	59.8	57.3	55.0	58.6	58.6	59.8	58.5	57.3	56.1	59.8
Alkalinity (CO_3)	(mg/L)	0.0	0.0	0.0	--	--	--	--	0.0	0.0	0.0	0.0	0.0	--	--	--
Alkalinity (OH)	(mg/L)	0.0	0.0	0.0	--	--	--	--	0 0	0.0	0.0	0.0	0.0	--	--	--
Dissolved Oxygen	(mg/L)	7.6	7.8	7.4	7.9	7.7	7.3	7.9	7.7	7.4	7.7	7.3	7.6	7.6	7.4	7.4
T.O.C.	(mg/L)	20.5	20.0	20.0	19.5	19.5	20.5	19.5	20.0	19.5	19.5	20.0	20.0	20.0	19.5	20.5
T.I.C.	(mg/L)	9.5	9.0	10.0	10.0	10.0	10.0	10.0	9.5	10.0	9.5	10.0	9.0	8.0	9.5	8.5
Turbidity	(NTu)	1.0	1.0	1.0	1.0	1.0	1.0	1.0	0.8	1.0	1.0	1.0	1.0	1.0	1.0	1.0
Kjeldhal Total-N	(mg/L)	0.6	0.6	0.7	0.6	0.7	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6
NO_3-NO_2-N (Diss.)	(mg/L)	0.02	0.02	0.02	0.02	0.01	0.03	0.02	0.01	0.02	0.02	0.03	0.01	0.03	0.02	0.02
P-Total	(mg/L)	0.024	0.022	0.024	0.022	0.024	0.027	0.027	0 023	0.032	0.028	0.022	0.018	0.021	0.030	0.029
Cyanide (Tot.)	(mg/L)	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
As (Tot.)	(mg/L)	0.002	0.004	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.001	<0.001	<0.001	<0.001	<0.001
Cd (Ext.)	(mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Cu (Ext.)	(mg/L)	0.01	0.05	0.04	0.01	0.01	0.01	0.03	0.01	0.01	0.02	0.01	0.03	0.01	0.01	0.01
Pb (Ext.)	(mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Ni (Ext.)	(mg/L)	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Zn (Ext. Dir.)	(mg/L)	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Ca (Ext.)	(mg/L)	22.5	21.4	22.4	22.9	23.1	22.6	22.7	22.0	23.0	24.1	22.8	23.8	22.6	23.2	23.8
Mg (Ext.)	(mg/L)	7.30	6.91	7.30	7.46	7.50	7.44	7.35	7.19	7.50	7.94	7.43	8.06	7.42	7.57	7.83
Hg (Tot.)	(µg/L)	<0.02	--	<0.02	<0.02	<0.02	0.02	<0.02	--	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
Sample Depth	(m)	1.4	2.0	0.8	0.1	0.1	1.3	2.1	0.2	1.0	2.9	0.4	0.2	0.4	1.8	2.1
Total Depth	(m)	2.8	2.7	1.9	3.0	3.5	2.3	2.9	2.0	2.5	3.2	2.2	1.3	2.9	2.4	2.9

APPENDIX 4: METAL AND CYANIDE CONCENTRATIONS IN RICE LAKE SEDIMENTS AND SAN ANTONIO MINE TAILINGS.

PARAMETER STATION	Hg (wet) µg/g	Hg µg/g	As µg/g	Cd µg/g	Cu µg/g	Pb µg/g	Ni µg/g	Zn µg/g	Fe %	Mn µg/g	CN µg/g
1	0.28	1.51	28.9	2.	636.	40.	56.	196.	3.08	280.	-
2	0.22	1.06	14.5	2.	456.	30.	56.	175.	3.29	330.	-
3	0.21	1.93	45.5	2.	1020.	45.	75.	252.	3.57	440.	-
4	0.23	1.36	29.0	2.	528.	35.	50.	196.	3.43	320.	-
5	0.22	1.06	33.1	2.	564.	35.	50.	210	4.20	460.	<1.
6	0.26	1.41	35.0	2.	660.	25.	37.	168.	2.73	210.	-
7	0.29	1.80	23.4	2.	528.	30.	56.	217.	3.64	270.	-
8	0.23	0.99	35.8	2.	828.	30.	62.	210.	3.22	340.	-
9	0.32	1.65	40.0	<1.	660.	35.	50.	217.	3.92	300.	-
10	0.29	1.30	25.5	2.	624.	35.	50.	210.	3.85	400.	<1.
11	0.08	0.11	18.3	2.	36.	20.	31.	70.	4.13	630	-
12	0.21	1.83	37.9	2.	840.	40.	62.	210.	3.22	340.	-
13	0.44	1.79	41.4	2.	552.	40.	50.	238.	4.69	450.	-
14	0.38	2.08	41.0	2.	828.	35.	56.	231.	3.78	320.	-
15	0.59	1.46	41.4	2.	396.	30.	62.	238.	5.39	650.	<1.
TAILINGS #1	0.11	0.14	14.	<1.	60.	10.	37.	84.	44.1	690.	2.
TAILINGS #2	0.28	0.37	11.	2.	60.	25.	75.	140.	46.9	770.	2.
TAILINGS #3	0.08	0.10	24.	2.	72.	50.	56.	322.	49.7	720.	11.

APPENDIX 5(a): GENERAL PHYSICAL MERISTICS AND MUSCLE TISSUE METAL CONCENTRATIONS; NORTHERN PIKE, MAY 1981.

Length (cm)	Weight (g)	Sex	Age (Yrs)	Muscle Tissue Metal Levels (µg/g) Hg	As	Cd	Cu	Pb	Ni	Zn
42.7	480	F	5	0.43	<0.02	0.2	1.8	0.5	<0.5	3.7
43.8	530	F	4	0.38	<0.02	<0.1	0.8	0.8	<0.5	4.6
42.6	590	M	4.	0.54	0.08	<0.1	1.3	0.5	<0.5	3.8
48.0	720	F	4.	0.83	0.02	<0.1	1.3	0.5	1.0	3.4
45.5	580	F	4	0.66	0.04	<0.1	1.5	0.8	<0.5	4.4
53.9	530	F	5	0.68	<0.02	<0.1	2.0	0.8	<0.5	4.6
53.7	650	M	4	0.55	<0.02	<0.1	1.5	1.0	<0.5	4.4
41.8	540	M	4	0.38	0.04	<0.1	1.8	0.8	<0.5	3.4
43.8	590	F	4	0.80	<0.02	<0.1	1.3	1.0	<0.5	3.8
40.3	450	M	3	0.46	<0.02	<0.1	1.2	1.0	1.0	3.1
39.2	480	M	4	0.43	0.02	<0.1	1.5	1.0	2.0	3.5
43.7	600	F	4	0.54	0.02	<0.1	1.2	0.8	1.0	3.6
41.3	510	M	4	0.46	0.02	<0.1	1.0	0.8	<0.5	3.3
47.5	670	M	4	1.12	<0.02	<0.1	0.9	1.0	<0.5	3.8
41.1	500	F	4	0.38	<0.02	<0.1	1.6	0.8	<0.5	3.5
39.0	460	M	4	0.50	<0.02	<0.1	0.9	0.8	<0.5	3.3
42.4	560	F	4	0.40	<0.02	<0.1	0.9	0.8	1.0	3.1
39.0	420	M	4	0.32	<0.02	<0.1	1.2	1.0	<0.5	3.7
44.2	580	M	5	0.75	0.06	<0.1	1.2	1.0	<0.5	3.5
48.4	750	F	4	0.88	<0.02	<0.1	1.3	0.8	<0.5	3.8
42.0	510	M	5	0.60	<0.02	<0.1	1.3	1.0	<0.5	3.3
45.3	680	M	5	0.83	0.04	<0.1	1.2	1.0	1.0	3.2
46.4	660	M	5	0.89	0.02	<0.1	0.9	1.0	<0.5	3.7
38.3	420	M	4	0.28	<0.02	<0.1	0.8	1.0	<0.5	3.6
43.0	580	M	4	0.59	0.08	<0.1	0.9	0.5	1.0	3.5
45.2	660	F	5	0.53	0.02	<0.1	0.9	0.5	<0.5	3.0
43.7	560	F	5	0.30	0.04	<0.1	0.2	0.5	<0.5	3.3
44.8	630	F	5	0.47	<0.02	<0.1	0.2	0.8	<0.5	3.3
45.5	600	F	4	0.44	0.02	<0.1	0.2	1.0	<0.5	3.1
45.4	610	F	4	0.57	<0.02	<0.1	0.3	2.5	<0.5	4.2
43.4	580	M	5	0.38	0.05	<0.1	0.5	1.0	<0.5	4.2
43.5	580	F	4	0.42	0.03	<0.1	0.2	0.5	<0.5	2.7
45.5	600	M	5	0.61	<0.02	<0.1	0.2	0.8	<0.5	4.3
43.4	620	F	4	0.46	0.02	<0.1	0.3	0.5	<0.5	3.3
47.0	780	F	5	0.46	0.03	<0.1	0.3	0.8	<0.5	3.1
39.4	400	F	4	0.37	0.03	<0.1	0.3	1.0	<0.5	3.7
46.3	680	M	5	0.69	0.04	<0.1	0.2	1.0	<0.5	3.2
48.0	480	M	5	0.37	0.02	<0.1	0.8	1.0	<0.5	3.4
47.7	760	F	6	0.43	<0.02	<0.1	0.2	1.0	<0.5	2.8
44.0	640	F	5	0.49	0.03	<0.1	0.2	0.8	<0.5	2.4

APPENDIX 5(b): GENERAL PHYSICAL MERISTICS AND MUSCLE TISSUE METAL CONCENTRATIONS; COMMON SUCKER, MAY 1981.

Length (cm)	Weight (g)	Sex	Age (Yrs)	Muscle Tissue Metal Levels (µg/g) Hg	As	Cd	Cu	Pb	Ni	Zn
51.2	2810	F	8	0.38	<0.02	<0.1	0.8	0.7	1.0	4.5
46.5	1280	F	6	0.26	<0.02	<0.1	0.5	0.7	<0.5	3.3
47.6	1340	F	6	0.22	<0.02	<0.1	0.4	0.5	<0.5	2.8
40.4	900	M	6	0.20	<0.02	<0.1	0.9	0.7	<0.5	4.4
51.1	1600	F	10	0.42	<0.02	<0.1	0.6	0.5	<0.5	2.8
44.5	1120	F	5	0.25	<0.02	<0.1	0.4	0.5	<0.5	3.0
44.6	1300	M	15	0.37	<0.02	<0.1	0.6	0.5	<0.5	3.9
42.7	920	M	15	0.29	<0.02	<0.1	0.4	0.7	<0.5	2.9

APPENDIX 5(c): GENERAL PHYSICAL MERISTICS AND MUSCLE TISSUE METAL CONCENTRATIONS; YELLOW PERCH, MAY 1981.

Length (cm)	Weight (g)	Sex	Age (Yrs)	Muscle Tissue Metal Levels (µg/g) Hg	As	Cd	Cu	Pb	Ni	Zn
28.7	330	M	6	0.24	<0.02	0.1	0.4	0.9	<0.5	5.4
28.2	380	F	6	0.24	<0.02	<0.1	0.5	1.1	<0.5	5.2
32.5	590	F	7	0.32	<0.02	<0.1	0.5	1.1	<0.5	5.1
29.8	400	M	7	0.44	<0.02	<0.1	0.4	1.2	<0.5	5.4
31.0	550	F	7	0.33	<0.02	0.1	0.4	1.2	<0.5	5.0
25.5	240	F	5	0.36	<0.02	0.1	0.6	0.5	<0.5	5.3
26.4	260	F	5	0.32	<0.02	<0.1	0.5	0.7	<0.5	4.9
24.3	190	M	4	0.24	<0.02	<0.1	0.4	0.7	<0.5	5.6
34.7	620	F	8	0.46	<0.02	<0.1	0.4	0.5	<0.5	4.4
29.2	410	F	7	0.36	<0.02	<0.1	0.4	0.7	<0.5	4.9
30.7	500	F	8	0.50	<0.02	<0.1	0.3	0.7	<0.5	4.5
32.7	480	F	8	0.46	<0.02	<0.1	0.6	0.5	<0.5	5.9
33.2	580	F	8	0.59	<0.02	<0.1	0.4	0.5	<0.5	3.9
29.2	370	M	7	0.54	<0.02	0.1	0.5	0.7	<0.5	4.4
25.7	250	F	5	0.21	<0.02	<0.1	0.4	0.5	<0.5	4.9
30.5	410	F	6	0.43	<0.02	0.1	0.4	1.1	<0.5	5.4
29.1	380	F	6	0.36	<0.02	0.1	0.4	1.1	<0.5	5.8
29.6	360	M	6	0.47	<0.02	0.1	0.3	0.7	2.0	4.2
27.8	330	F	5	0.23	<0.02	<0.1	0.6	0.7	<0.5	5.3

APPENDIX 6: ESTIMATED DISTANCES (m) TO RICE LAKE SAMPLING STATIONS.

STATION	ESTIMATED DISTANCE (m) TO: NEAREST POINT ON TAILINGS	MINE SHAFT DEWATERING OUTFALL
1	1210	1560
2	1100	1810
3	1210	1420
4	1460	1950
5	870	1290
6	950	960
7	580	710
8	1350	1970
9	600	630
10	1230	1780
11	10	500
12	1000	1170
13	430	550
14	830	980
15	280	710